Ecological Role of Mining Ponds in Southern Coastal Mines, Namibia

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MRTLIE006

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15 April 2020

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I know the meaning of plagiarism and declare that all of the work in the thesis, save for that which is properly acknowledged, is my own.

I hereby declare that the work presented in this thesis is my own. I have not used any other sources than the ones stated in the text, which are clearly indicated and acknowledged.

Signature

Signed by candidate

Liezl Maritz

Date: 15 April 2020

ACKNOWLEDGEMENTS

This thesis is dedicated to my husband and parents for your continual prayers, patience and support throughout this journey, I appreciate you.

I would like to express my gratitude to my supervisors Prof Branch and Dr Pillay, for your guidance, motivation and sharing of your immense knowledge. I would like to give special thanks to Prof Colin Attwood for assisting me with the otolith analysis and Dr Christian Prins who introduced and assisted me with the statistical programmes used in my project. Prof Janine Adams and Dr Taryn Ridden supplied data on saltmarshes associated with the Orange River, and Prof Ladislav Mucina kindly identified the saltmarsh species growing around the ponds. Dr Rene Navarro provided access to data on CWAC bird counts for the Orange River.

Lastly, I would like to thank Namdeb Diamond Corporation (Pty) Ltd for funding my MSc and allowing me to conduct this study in the Southern Coastal Mines licence area.

ABSTRACT

Targeting marine diamondiferous deposits along the south-western Namibian coastline has involved the construction of seawall berms to advance the coastline and permit mining in previously subtidal areas. Large areas are mined out to bedrock level by the removal of overburden (sand and gravel), and after mining is complete, areas behind the seawalls fill with seawater, creating a series of coastal marine ponds that have the potential to function as saline wetlands corresponding to closed estuaries.

The study site lies north of the Orange River mouth, within the Tsau//Khaeb (Sperrgebiet) National Park, to which the public has restricted access due to diamond mining in the area. Consequently, there is a focus on biodiversity conservation and the overall objective of this thesis was to determine the ecological value of the mining ponds by investigating whether they harbour sufficient biodiversity to qualify as a viable alternative ecological habitat, rather than restoring them to their original state as backfilled, revegetated dune areas.

The study area, approximately 75 kilometres in length, was divided into north, middle and south sections, according to the age and status of mining activities, and I first assessed whether ponds in these three areas differed in their physical properties (Chapter 1). To determine whether the ponds serve as a useful ecological role worth preserving, I then investigated the diversity and amount of saltmarsh vegetation (Chapter 2), bird species (Chapter 3) and fish (Chapter 4) currently benefiting from these mining ponds.

Ponds in the north are older and hypersaline whereas the south and middle ponds are younger and closely approach physical conditions in the sea. Diurnal fluctuations in oxygen concentration took place, but oxygen levels never dropped below 80% and were thus not limiting. After about 15 years, ponds developed salinities in excess of 80‰, which is likely to set limits on their ecological viability.

Most ponds supported saltmarshes, but only a single species, *Salicornia natalensis*, grew around them. Its abundance was greatest around old ponds, but its health decreased with salinity. Wind emerged as a likely means of dispersal among ponds.

The ponds supported a rich avifauna, averaging 11028 birds per count for all ponds combined; 36 species were recorded, ten being endemics, and five being listed in Namibia's Red Data Book. Numbers were highest for ponds that were being 'dewatered' to remove water prior to mining, as this exposed a rich benthic epifaunal source of food. Blacknecked Grebe, Cape Cormorant, Greater and Lesser Flamingos, Kelp Gull and Common Tern were the most abundant birds. Salinity did not limit bird numbers or diversity, so the northern high-salinity ponds may continue to serve as bird habitat for periods of time much greater than the 15 years after which they become hypersaline. In comparison with nine other wetlands in the region, the ponds had great numbers, diversity, densities, and numbers of threatened species than about half of these wetlands, many of which are considered Important Bird Areas (IBAs). In many cases, they also supported more species that had numbers in excess of 1% of the southern African population than these IBAs.

Only two species of fish commonly occurred in the ponds, the west coast steenbras *Lithognathus auratus* and the southern mullet *Chelon richardsonii*, although small numbers of two other marine species were recorded. The diversity of marine fish was thus low, even by the impoverished standards of west-coast estuaries. Unexpectedly, there were no significant differences between the ichthyofauna of ponds in the north, middle and south, nor was there any relationship between total fish numbers and salinity. Body condition of steenbras was lowest in the hypersaline north ponds, and their stomach contents contained a low diversity of food items there. The presence of recruits and the range of gonadal states suggested that both species bred in the ponds. Steenbras proved to be protandrous, with females dominating larger size classes. Growth rates of both species were faster in the ponds than in the adjacent sea, and mullet achieved substantially greater sizes in ponds.

Thus, the ponds do serve as viable ecosystems, albeit with a limited range of saltmarsh and fish species, and support an impressively diverse avifauna. Their long-term viability will, however, become limited by rising salinity as their age increases.

GENERAL INTRODUCTION AND OVERVIEW OF THESIS

Alluvial diamonds have been mined in south-western Namibia since 1907, with the current mining licences belonging to Namdeb Diamond Corporation (Pty) Ltd, which holds eight mining licences (Simmons 2005). The area in which my study falls is located in the Southern Coastal Mines, the southernmost mining license area known as ML43 (Fig. 1).

Southern Coastal Mines is Namdeb's largest mining licence area, spanning both onshore and offshore, and extending for approximately one hundred kilometres along the coast north of the Orange River mouth. Targeting diamondiferous deposits along the coastline has involved the construction of a series of seawall berms to advance the coastline into the sea and permit mining along previously subtidal areas of the coast. Large areas previously lying below the low-water mark are thus exposed and are then mined out to bedrock level by the removal of overburden material (soil and rock lying above ore body), and later become filled with seawater, creating a series of large marine ponds.

Over the years, this opencast mining method has severely impacted a significant strip of land inside the mining licence area, which is situated within the Tsau //Khaeb (Sperrgebiet) National Park (Fig. 1). The original coastline has been completely altered and the marine ponds that are left behind now cover large areas, and have persisted for many years.

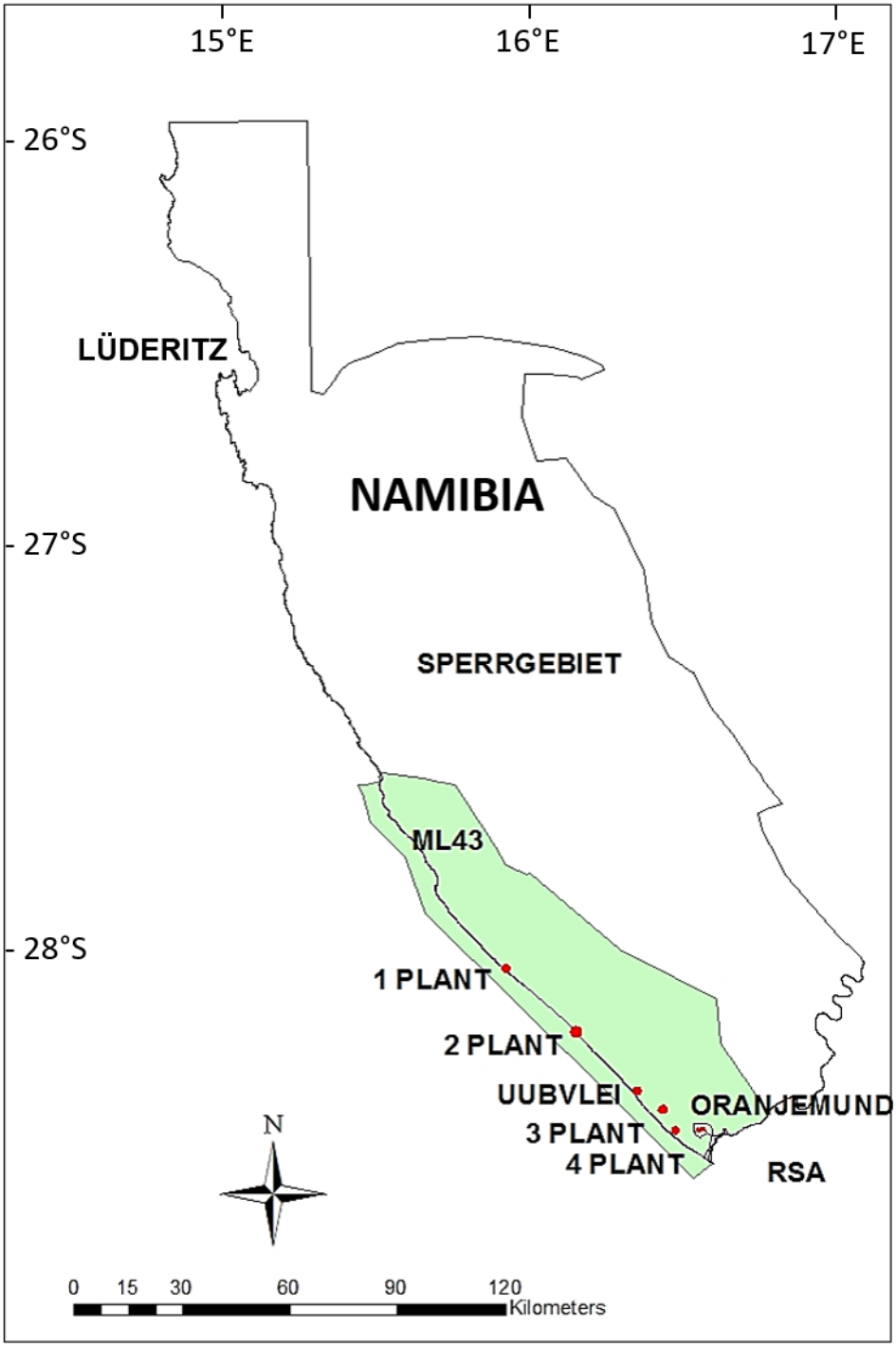

Figure 1: Location of Southern Coastal Mines Licence Area (ML43 shaded green) within the Tsau //Khaeb (Sperrgebiet) National Park in Namibia

PURPOSE OF STUDY

Since Namdeb is operating in a national park there is a focus on biodiversity conservation. Namdeb's current rehabilitation closure plan for the area includes backfilling of some ponds in Southern Coastal Mines along the access road that is more than 500 meters from the coastline. Rehabilitation could be employed to fill these ponds after mining, but it is possible that these marine ponds generate unique habitats along the coast that serve as habitat for plants and macrobenthos associated with saline wetlands, attract birdlife, and in some cases support fish populations. In addition, some ponds have been used for mariculture, particularly for oysters.

The ecological value of these ponds has never been determined, and my thesis probes various aspects of their ecology to fill this void. A study of the nature of these ponds might challenge the requirement for rehabilitation of all ponds back to a state resembling the original condition of the environment, if it provides evidence that the ponds harbour enough biodiversity to qualify as an alternative habitat that enhances biodiversity, rather than being 'rehabilitated' into backfilled, terrestrial, revegetated areas. The option therefore exists to retain some or all of these features; but the advisability of this depends very much on the physical and biological characteristics of the ponds. Rehabilitation will also be extremely costly. The research for this thesis therefore explores the potential ecological benefits the mining ponds hold for the region, and whether retention of ponds would be sufficiently beneficial to obviate the need for their rehabilitation.

The thesis was developed in four phases, each comprising a chapter:

a) Chapter 1: A survey of the ponds in terms of their position, size and dimensions, and an assessment of their physical conditions both across a series of ponds from North to South, and within selected ponds over a 24-hour period to explore daily changes.
b) Chapter 2: Quantification of the diversity and amount of saltmarsh vegetation associated with the ponds to determine if the ponds provide suitable habitat on a coast in which wetland habitat is largely absent because of the arid nature of the environment. Specifically, the chapter will probe whether the ponds can act as 'stepping stones' for the distribution of saltmarsh plants along the coast.
c) Chapter 3: A survey of the use of the ponds by birds, and a comparison of the avifauna in them with published accounts of other wetlands, particularly the Orange River

Estuary avifauna. In short, the goal is to determine whether the ponds constitute a significant habitat for birds.

d) Chapter 4: Quantification of the fish fauna of the ponds, including identification of the species present, assessment of their relative abundance and diversity, and the sizes, body condition, diets, and gonadal cycles and growth rates of selected species.
e) Chapter 5: a brief synthesis of the findings of the previous chapters, with an assessment of the ecological importance of the ponds.

A separate study will be conducted by another student (J. Cloete) to determine whether the benthic invertebrate fauna is sufficient to support viable bird and fish populations in the ponds. As this benthic research was not yet complete at the time I wrote my thesis, I drew on the information in a non-quantitative manner to help explain patterns emerging in the bird and fish life. The overall focus of my study will fall on the ecological viability of the ponds in Southern Coastal Mines and will adopt an ecosystem approach.

REFERENCES:

Simmons RE. 2005. Declining coastal avifauna at a diamond-mining site in Namibia: comparisons and causes. Ostrich 76: 97-103

CHAPTER 1: Background Survey of the Physical Characteristics of the Mining Ponds

OVERVIEW OF MINING PONDS

Ponds of considerable size are created during coastal marine mining for diamonds in southern Namibia. The process of mining relies on accretion of the shore by seawalls that push the coast seawards and allow mining of previously subtidal diamondiferous sediments (summarised schematically in Fig. 1.1), generating seawalls and ponds that lie parallel to the coast.

These operations take place on the Atlantic coast and fall within the Benguela Ecosystem – one of the world's four main eastern-boundary upwelling systems (Hutchings et al. 2009). The distance between the most southern and the most northern mining ponds is approximately 75 kilometres. The study area was divided into three sections (Fig. 1.2) namely south (~15 kilometres), middle (~10 kilometres) and north (~50 kilometres) based on the age of the ponds and the status of mining activities within each section.

Namdeb is actively mining in the southern and middle areas, slowly progressing towards the north. During the time of the study, a number of southern ponds were being 'dewatered' and actively mined. Other ponds there have stabilised post-mining, but they too are relatively young. In the middle area where active mining has occurred recently or is still occurring, some of the ponds are very young – the youngest and least mature in the region. The northern ponds, from Uubvlei onwards have persisted for many years since all mining activities in that area came to an end between 1990 (1 Plant areas) and 2006 (2 Plant areas) so the ponds in that section are older and more mature.

The mining ponds are scattered along the coastline, often >500 m from the original coastline, with some ponds lying beyond and others within the original shoreline (see white line in Fig. 1.2).

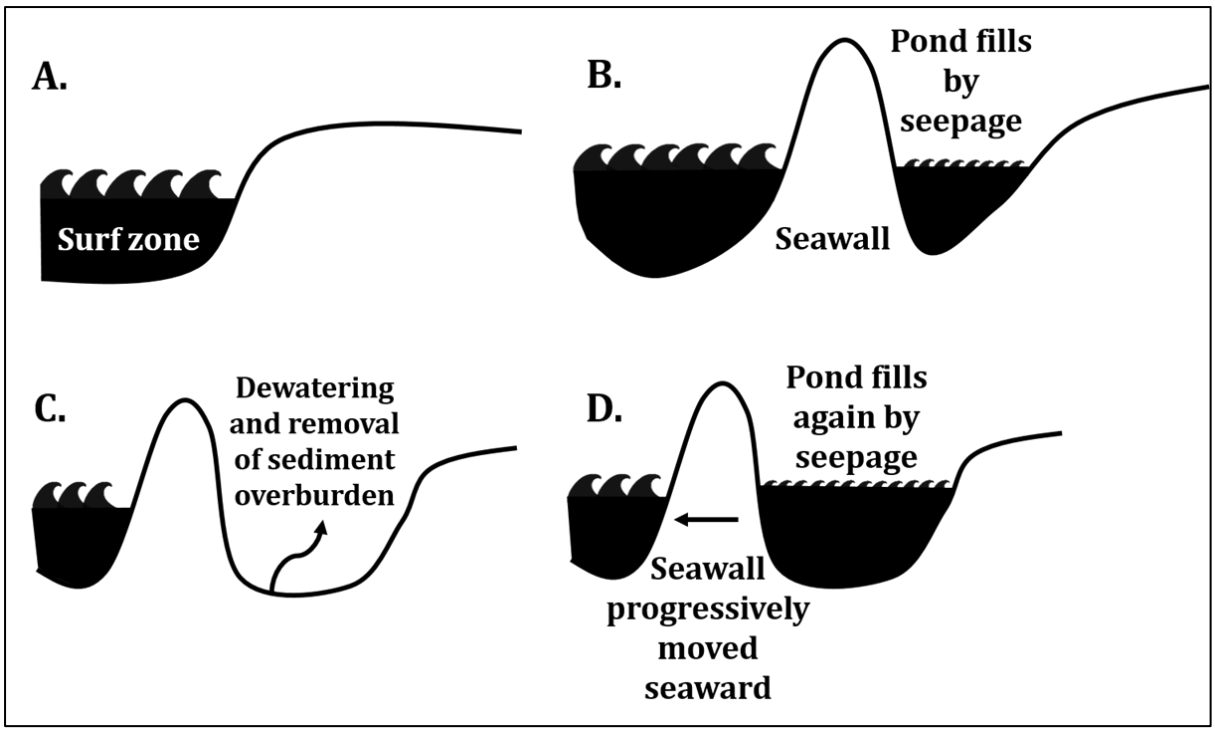

Figure 1.1: Schematic diagram illustrating the process of accreting the shoreline seawards to allow mining in the previously subtidal surf zone, and the formation of ponds on the landward side.

Figure 1.2: Study areas (28.16S, 15.89°E to 28.60°S, 16.40°E), with the white line indicating the original coastline. Map derived from Google Earth.

Southern and middle ponds

Current mining in the southern and middle areas has adopted mining by accretion, using one of three methods. The first method is to use a hydraulic suction dredge to pump overburden sedimentary material from the ponds onto the beach, resulting in deposition of sediments that pushes the shoreline seawards (Fig. 1.3). The second method is to use earthmoving machines to remove and transport overburden material from the mining sites and old tailing dumps onto conveyor belts, which then dump the material onto the beach and into the sea, again assisting accretion (Fig. 1.4). Lastly, a new accretion technique has recently been applied called 'beach nourishment' where overburden material is dumped in heaps over an extended area in front of the seawall to create a man-made beach (Fig. 1.5).

In all cases, wave action then distributes the sand onto the beach, developing a barrier (or seawall berm), landward of which marine ponds form behind the seawall berm, fed by seepage into the ponds situated closest to the seawall. The creation of successive seawalls progressively pushes back the sea, allowing Namdeb to access the previously subtidal areas that were not accessible to them in the old breaker zone. Accretion allows Namdeb to continuously move the seawall out to sea. Opencast dry-mining then takes place on these accreted beaches behind the seawalls. Advancement of the seawalls shifts them progressively seawards, away from the ponds created. Eventually the majority of the southern ponds will be disturbed either by dredgers pumping the water and overburden out to sea or, when mining takes place, the entire pond has to be dewatered. After mining has been completed, the ponds do, however, refill and life becomes established in them once again.

Figure 1.3: Dredger in pond (left) enlarged and showing pipeline (middle) and transport of suspended sediments (right) from the dredger to where they are deposited on the shore.

Figure 1.4: Accretion Conveyors (left) transporting sediments that are deposited on the shore (middle) to assist with accretion (right).

Figure 1.5: Beach Nourishment (left), in which sediment is deposited on the shore (middle) to progressively accrete the shoreline and add to seawalls (right).

Northern ponds

Previously, mining in the northern areas extended only up to the shoreline. No accretion methods were applied to push the sea back to open additional mining areas, although seawalls were constructed to protect the mining areas. Once mined out, the areas, which then lay below sea level, were abandoned and ponds filled up naturally through seepage.

Due to their earlier development, a number of these northern ponds are relatively shallow because of evaporation and because they have been filled in by wind-transported sediments, and can be expected to have higher salinity levels that create unfavourable

conditions for most ecological processes. The northern ponds characteristics may therefore give an indication of how the southern and middle ponds can be expected to look sometime after they have been abandoned after mining.

I hypothesised that there would be significant spatial differences between the ponds' water quality parameters in the different areas due to their age and depth differences. More specifically, because of the greater age of the north ponds and their resultant shallower depths, I expected the north ponds to be warmer and more saline than the south and middle ponds, but that pH, oxygen levels and chlorophyll concentrations would not necessarily differ among the three regions. I also hypothesised that temperatures and salinities in the ponds would be greater than in the sea due to their shallowness and evaporation, that pH would be higher in ponds as the pH in the Benguela Upwelling Ecosystem is naturally low, and that oxygen in the ponds would be higher than in the sea because upwelling in the ocean regularly introduces oxygen-deficient water to the coast (Chapman and Shannon 1985).

In addition to these spatial differences in variables, I anticipated there would be 24-hour cycles for some of the variables. I expected that (a) temperature would rise during the day due to solar heating and would span a substantial range because of the shallowness of the ponds; (b) oxygen concentrations would increase by day due to photosynthesis and drop by night due to respiration; (c) pH would follow a similar but more damped pattern, but that (d) chlorophyll would change little and (e) salinity virtually not at all.

MATERIALS AND METHODS

Study area

The study site lies north of the Orange River mouth, the longest river in southern Africa and one of few perennial rivers on the arid south-west African coast (Simmons and Allan 2002). The Atlantic Ocean lies parallel to Southern Coastal Mines (28°S) and plays an important role in the ponds' water quality.

Like natural closed estuaries, the mining ponds undergo changes of temperature, dissolved oxygen, salinity, pH and chlorophyll depending on evaporation rates, rainfall

(which is minimal) and, most importantly, seawater inputs when seawater overtops the seawall, enters through seepage, or when there are periodic occurrences of breaching (Fig. 1.6).

Figure 1.6: An example of a seawall breach. Satellite image derived from Google Earth.

Sampling methods

A once-off spatial survey was conducted by using Google Imagery to identify all the mining ponds along Southern Coastal Mines coastline to establish their number, position, size and accessibility. To ground-truth this information and to obtain information on the physical conditions of the ponds, field visits were conducted to all accessible ponds; and for more detailed and intensive studies, measurements were made in a subsample of ponds in November 2017 (Figs. 1.7, 1.8 and 1.9). The study area was divided into three sections according to the current mining activities, namely south (S: with current dewatering and active mining), middle (M: active and recently mined areas) and north (N: no recent mining activities). Of the 150 mining ponds, 24 are located in the south, 15 in the middle and 111 in the north. Measurements were made at a water depth of 60cm in 33 representative ponds (11 per area) and in the ocean, to determine temperature,

salinity, dissolved oxygen, pH and chlorophyll-*a* levels (Chl). Measurements were undertaken using a YSI 650 Multi-parameter Display System and following the approaches advocated by Adams (2013). Two sets of measurements were made. The first was a spatial comparison done during spring (November 2017), to determine how water quality differed among the 33 representative ponds, and was done at a consistent period around midday (10h00-14h00) to minimise diurnal effects. For logistical reasons, only single measurements were taken in the ocean, so those data are not employed to make statistical comparisons with the ponds, but are uses for qualitative comparisons.

The 33 representative ponds, 11 per area, were as follows:

1. South (S) ponds 1-24:
   S4, S5, S6, S7, S10, S15, S16, S17, S20, S22, S23
2. Middle (M) ponds 25-39:
   M25, M26, M27, M28, M29, M30, M31, M32, M33, M35, M37
3. North (N) ponds 40-150:
   N42, N44, N49, N56, N100, N106, N109, N115, N128, N132, N138

Secondly, three representative ponds per area (S3, S4, S23; M26, M30, M37; N42, N44, N49) were measured for the same variables over a period of 24-hours in November 2017 to determine the magnitude of diurnal variations within the mining ponds.

Figure 1.7: North Ponds. Map derived from Google Earth. (Green icons illustrate 11 selected ponds; red asterisk show ponds monitored over 24-hours).

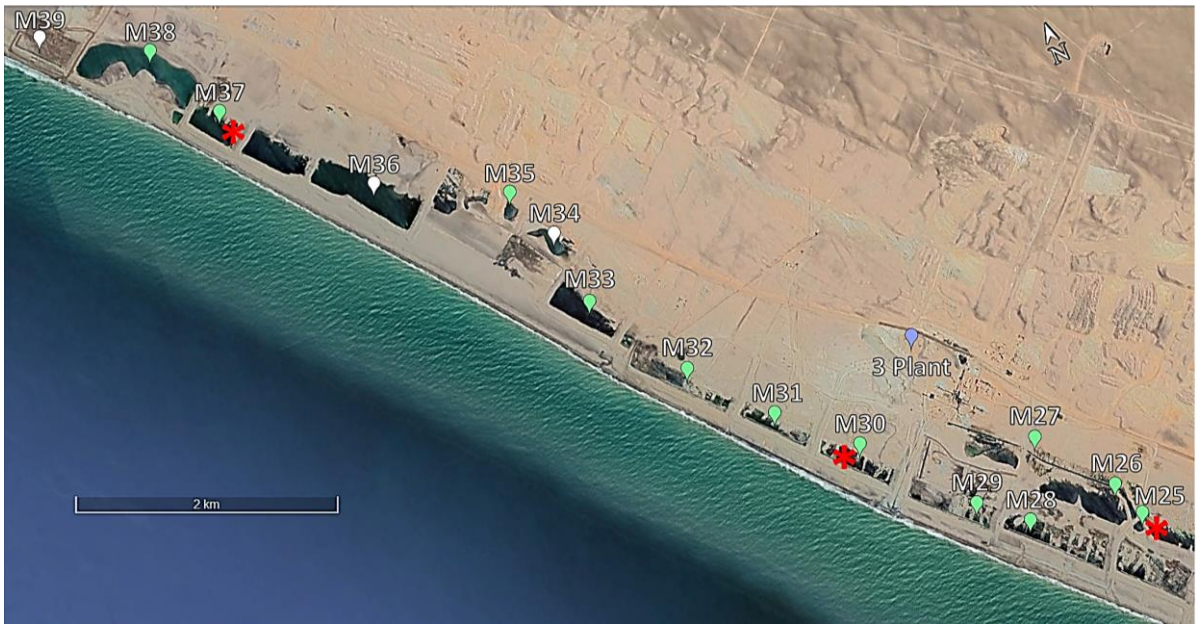

Figure 1.8: Middle Ponds. Map derived from Google Earth. (White icons illustrate the location of a mining pond, green icons illustrates the 11 selected ponds; red asterisk show ponds monitored over 24-hours).

Figure 1.9: South Ponds. Map derived from Google Earth. (White icons illustrate the location of a mining pond, green icons illustrates the 11 selected ponds; red asterisk show ponds monitored over 24-hours).

The 33 representative ponds varied in sizes and were classified as big (> 0.5 km^2), intermediate (0.16–0.5 km^2) and small (< 0.16 km^2) (Fig. 1.10), but as none of the variables explored was related to pond size, size was not employed as an explanatory variable.

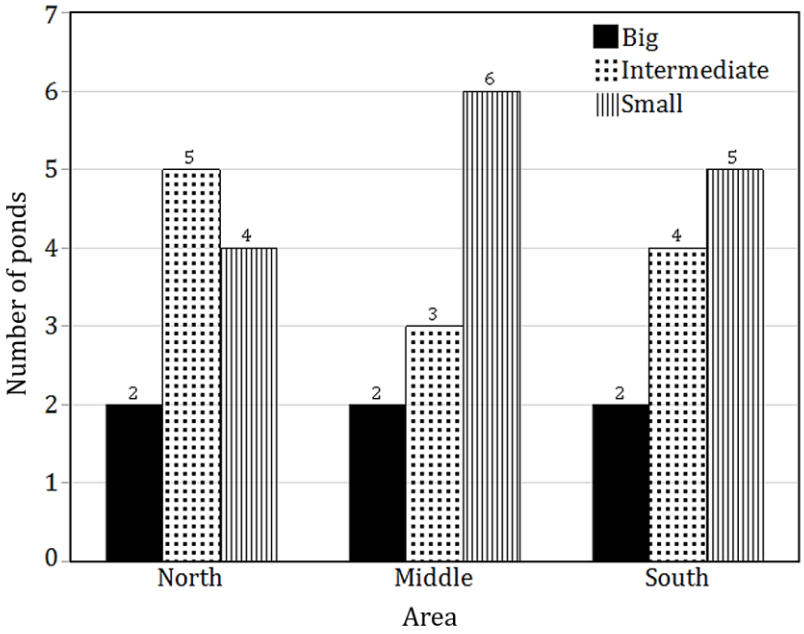

Figure 1.10: Sizes of ponds sampled in North, Middle and South areas

Statistical analysis

Data were analysed using statistical software JMP (version 14) developed by the Statistical Analysis System Institute (SAS).

One-way analysis of variance (ANOVA) with a significance level of 0.05 was used to determine significant differences in each water parameter among the north, middle and south areas. The results were expressed as F ratios and p-values. When significant differences among means were found, post-hoc Tukey honestly significant difference (HSD) tests were run to determine the significance of differences among individual areas. Data for the open sea adjacent to the ponds are included for comparison, but not incorporated in the statistical comparisons, as they comprised single measurements.

For the 24-hour samples, I again used one-way ANOVA (and post-hoc Tukey tests) to compare among areas and ponds, as a means of verifying whether these data accorded with those arising from the spatial survey. Best-fit curves were fitted to the 24-hour data, but I did not apply any statistical tests to those data because it was impossible to sample all the ponds at simultaneous times, and I was in any case seeking patterns rather than statistical significance, and the magnitudes of changes did not form part of my *a priori* hypotheses.

To explore relationships among variables, I used a Pearson's correlation (r) between oxygen and pH (for the 24-hour data series), and determined best-fit regressions and coefficients of determination (R^2) between age of ponds and salinity, and between salinity and chlorophyll levels (for the spatial comparison of 33 ponds). I used time-since-mining as a proxy for the age of the ponds, and excluded three ponds because they had periodically been breached, establishing marine conditions and effectively reducing their age-since-establishment. I excluded one outlier from the salinity vs. chlorophyll correlation – although the relationship remained strongly significant ($p < 0.001$) even if it was included.

RESULTS

Spatial comparisons:

The spatial comparisons among the 33 representative ponds and the ocean's water measurements are summarised in Table 1.1 and Fig. 1.11.

One-way analyses of variance revealed significant differences among areas for three of the variables: temperature ($F_{2,30} = 14.65$, $p < 0.0001$); salinity ($F_{2,30} = 31.12$, $p < 0.0001$); chlorophyll ($F_{2,30} = 4.07$, $p = 0.027$). Differences were not significant for pH and dissolved oxygen ($p > 0.05$ in both cases).

Table 1.1: Mean water quality parameters for the 33 representative ponds and the ocean

Pond No.	GPS Coordinates	Size	Last Mined	Temp. (°C)	Salinity (‰)	pH	Chl-*a* (µg/L)	DO (%)
South 4	28°35'31.34"S 16°23'28.75"E	Intermediate	December 2014	21.76	37.39	8.15	0.8	91.6
South 5	28°35'9.50"S 16°23'1.24"E	Intermediate	April 2015	23.04	36.87	8.07	1.4	99.1
South 6	28°35'5.96"S 16°22'36.78"E	Intermediate	January 2017	21.21	36.3	8.26	10.4	107.8
South 7	28°34'47.84"S 16°22'31.39"E	Intermediate	July 2016	22.11	32.51	8.95	49.9	125.9
South 10	28°34'36.24"S 16°21'45.25"E	Small	May 2017	20.33	35.57	8.37	2.4	111.8
South 15	28°33'17.67"S 16°20'11.76"E	Big	May 2012	20.05	39.48	7.84	1.5	97.7
South 16	28°33'3.49"S 16°19'54.86"E	Small	September 2007	21.01	37.76	8.19	6.6	112.9
South 17	28°32'20.71"S 16°19'14.75"E	Small	January 2010	22.13	38.02	8.09	3.6	103.6
South 20	28°31'43.16"S 16°18'28.90"E	Small	July 2016	18.75	40.58	7.9	26.7	94.2
South 22	28°31'20.40"S 16°18'16.42"E	Small	July 2016	21.07	39.23	8.09	11.1	101.6
South 23	28°31'8.45"S 16°18'7.49"E	Big	December 1993	22.6	39.66	8.31	5.6	105.2
Middle 25	28°30'39.13"S 16°17'23.88"E	Intermediate	June 2018	20.3	39.21	8.17	17.6	105.7
Middle 26	28°30'29.76"S 16°17'22.26"E	Small	December 1980	18.67	39.29	8.16	72	95.6
Middle 27	28°30'11.68"S 16°17'5.04"E	Small	December 1980	19.02	35.99	8.05	12.5	98.8
Middle 28	28°30'25.06"S 16°16'54.08"E	Small	August 2017	16.76	35.55	7.91	24.2	97.7
Middle 29	28°30'17.78"S 16°16'44.98"E	Intermediate	November 2016	16.7	40.31	7.92	41.4	99.4
Middle 30	28°30'0.21"S 16°16'22.58"E	Intermediate	December 2016	20.4	35.31	8.4	19.3	149.2
Middle 31	28°29'39.91"S 16°16'1.48"E	Big	January 2016	18.04	35.3	8.17	1.4	93.7
Middle 32	28°29'23.41"S 16°15'44.53"E	Big	November 2015	19.54	41.22	7.97	1.2	96.5
Middle 33	28°29'3.62"S 16°15'30.21"E	Small	December 1997	20.64	72.16	8.14	77.1	99.3
Middle 35	28°28'24.92"S 16°15'17.99"E	Small	December 1997	24.33	49.19	8.31	8.2	108.5
Middle 37	28°27'32.58"S 16°14'12.06"E	Small	August 2016	20.51	36.99	8.48	6.3	135.3
North 42	28°26'31.75"S 16°13'23.64"E	Intermediate	August 2006	21.85	45.62	7.95	0.9	110.3
North 44	28°26'2.39"S 16°12'55.17"E	Intermediate	February 2006	21.91	70.77	8.24	2.1	109.2
North 49	28°24'43.01"S 16°11'33.66"E	Big	June 2004	21.43	70.79	8.21	8.1	102.4
North 56	28°23'52.82"S 16°10'41.77"E	Big	December 2002	22.96	102.23	8.07	0.3	117.3
North 100	28°17'11.99"S 16° 3'21.60"E	Intermediate	December 2002	23.25	80.58	8.16	2.7	103.7
North 106	28°16'11.32"S 16° 2'15.76"E	Intermediate	December 2002	24.63	46.17	8.75	14.4	134.2
North 109	28°15'55.87"S 16° 1'55.48"E	Small	December 2002	24.97	88.97	8.66	15.6	136.5
North 115	28°15'21.11"S 16° 1'10.93"E	Small	December 2002	24.18	110.89	8.26	0.1	112.9
North 128	28°13'52.07"S 15°59'8.62"E	Intermediate	December 2002	22.1	99.61	8.15	0.7	109.1
North 132	28°12'51.73"S 15°57'54.77"E	Small	December 2002	24.03	109.55	8.23	0.1	108.1
North 138	28°11'32.31"S 15°56'14.35"E	Small	December 2002	24.12	116.37	8.19	0.8	124.3
Ocean	N/A	N/A	N/A	12.38	34.98	7.68	3.3	94.5

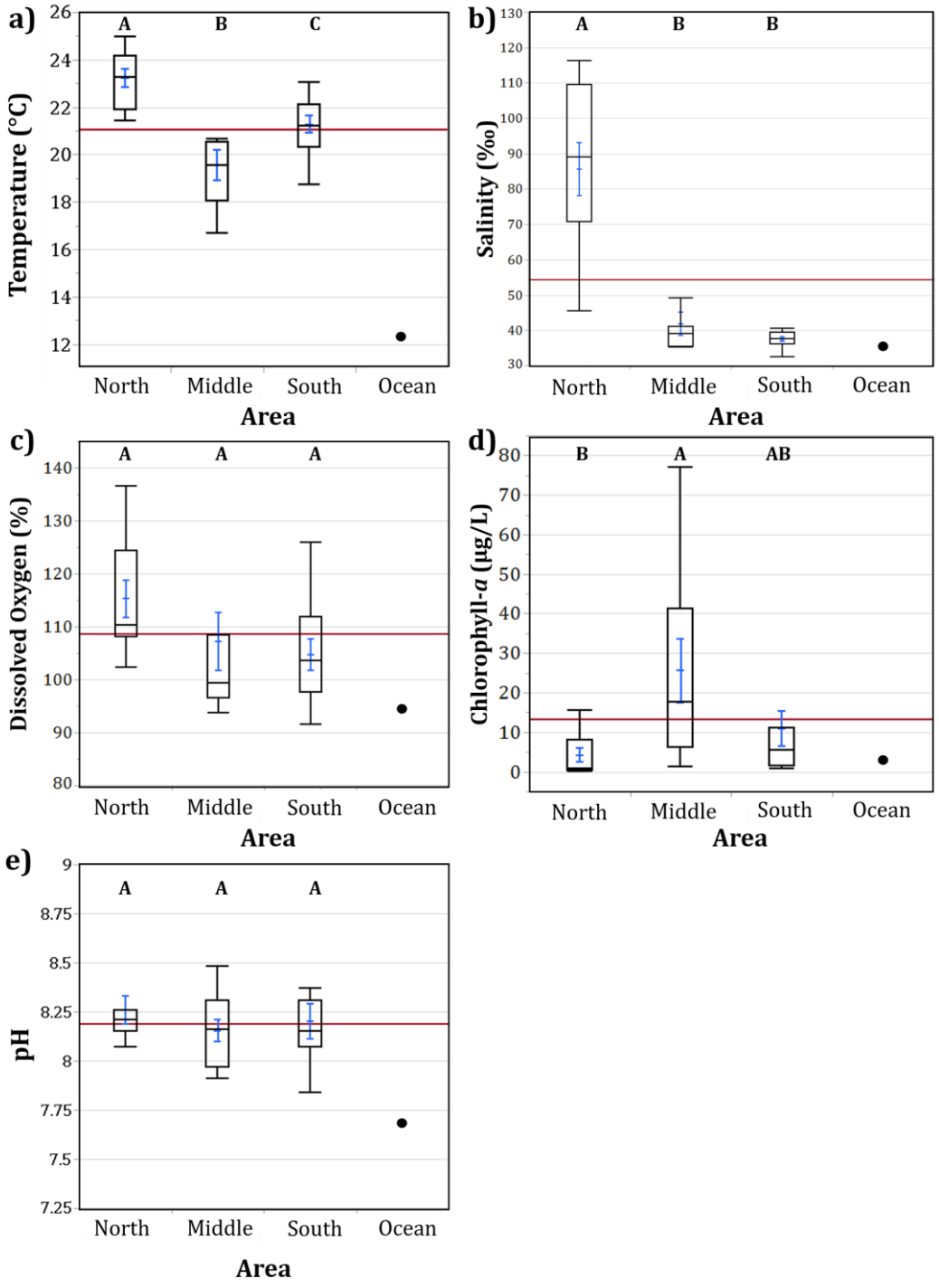

Figure 1.11: Water quality parameters: means of the 33 representative mining ponds (N = 11 per area) and the ocean (N = 1) measured once off for a) temperature, b) salinity, c) dissolved oxygen, d) chlorophyll-*a*, e) pH. The red lines show the grand mean and the blue whiskers inside the quartile box display the mean error bars. Black box and whisker plot show median values. Similar uppercase letters inside the graphs denote homogeneity between areas as calculated by post-hoc Tukey tests.

Post-hoc tests showed that temperatures in ponds were substantially and significantly greater in the north than either middle or south ponds, which were also significantly different from each other (Table 1.2); and the sea temperature was markedly lower by 7-10°C than in any of the ponds (Fig. 1.11a). Salinity in the north ponds was significantly elevated over that in the remaining ponds (Table 1.3), which approximated that of the open sea (Fig. 1.11b). Chlorophyll-*a* levels were greater in middle than north ponds but no other significant differences existed among areas (Table 1.4), and concentration in the ocean overlapped with the variance of ponds in all three areas (Fig. 1.11d) but had a lower value than the means of those ponds.

Measures of pH did not differ significantly among the pond areas ($F_{2,30} = 0.54$, $p = 0.58$), but all ponds had values about 0.5 pH units above that of the sea (Fig. 1.11e). There were also no significant differences in dissolved oxygen among the ponds ($F_{2,30} = 1.81$, $p = 0.18$), despite apparently greater levels in the north, and although the level in the ocean was less than that in any of the three areas of ponds, the variance exhibited in middle and south ponds spanned the level in the sea (Fig. 1.11c). Post-hoc tests were not pursued for pH or oxygen concentrations in view of the absence of any significant main effects among areas.

Table 1.2: Post-hoc Tukey test for differences in temperature among areas

Area	**n**	**Mean**	**Standard Error** *(pooled estimate of error variance)*	**Post-hoc Tukey Tests:**		
North	11	23.22	0.48	A		
Middle	11	19.54	0.48			C
South	11	21.28	0.48		B	

NOTE: Means sharing a common letter in each of Tables 2-4 are not significantly different at a 0.05 level

Table 1.3: Post-hoc Tukey tests for differences in salinity among areas

Area	**n**	**Mean**	**Standard Error**	**Post-hoc Tukey Tests:**	
North	11	85.60	4.76	A	
Middle	11	41.87	4.76		B
South	11	37.58	4.76		B

Table 1.4: Post-hoc Tukey tests for chlorophyll among area

Area	n	Mean	Standard Error	Post-hoc Tukey Tests:	
North	11	4.07	5.44		B
Middle	11	25.56	5.44	A	
South	11	10.91	5.44	A	B

24-hour Monitoring:

The water parameter means obtained from the 24-hour measurements are shown in Table 1.5. Similar to the results for the spatial comparisons, the temperature means of the nine ponds differed very little among one another, with the lowest temperature mean being 18.5°C in M30 and the highest 20.9°C in S3.

Table 1.5: Water parameter means for the 9 ponds measured over 24-hours (S = South; M = Middle; N = North)

Sites	N42	N44	N49	M26	M30	M37	S3	S4	S23
Temperature (°C)	19.18	20.33	19.86	18.93	18.5	18.65	20.9	20.22	19.3
Salinity (‰)	110.04	71.2	103.71	38.65	35.57	37.44	40.78	37.58	41.59
Dissolved Oxygen (%)	100	101	104	104	126	119	104	99	101
Chlorophyll-*a* (µg/L)	1.21	4.51	1.19	69.97	13.23	12.52	0.61	1.69	3.82
pH	7.78	8	7.84	8.16	8.11	8.22	8.06	8.05	8.16

The lowest individual temperature recorded was 16.71°C (in M30) and the highest 23.34°C (in S3) (Fig. 1.12). The temperature in each pond started to peak late morning through to noon. After noon, the temperatures slowly dropped again through to midnight, with minima occurring in the early morning. The maximum daily ranges in each area were 3.6, 4.3 and 5.4°C in the north, middle and south respectively.

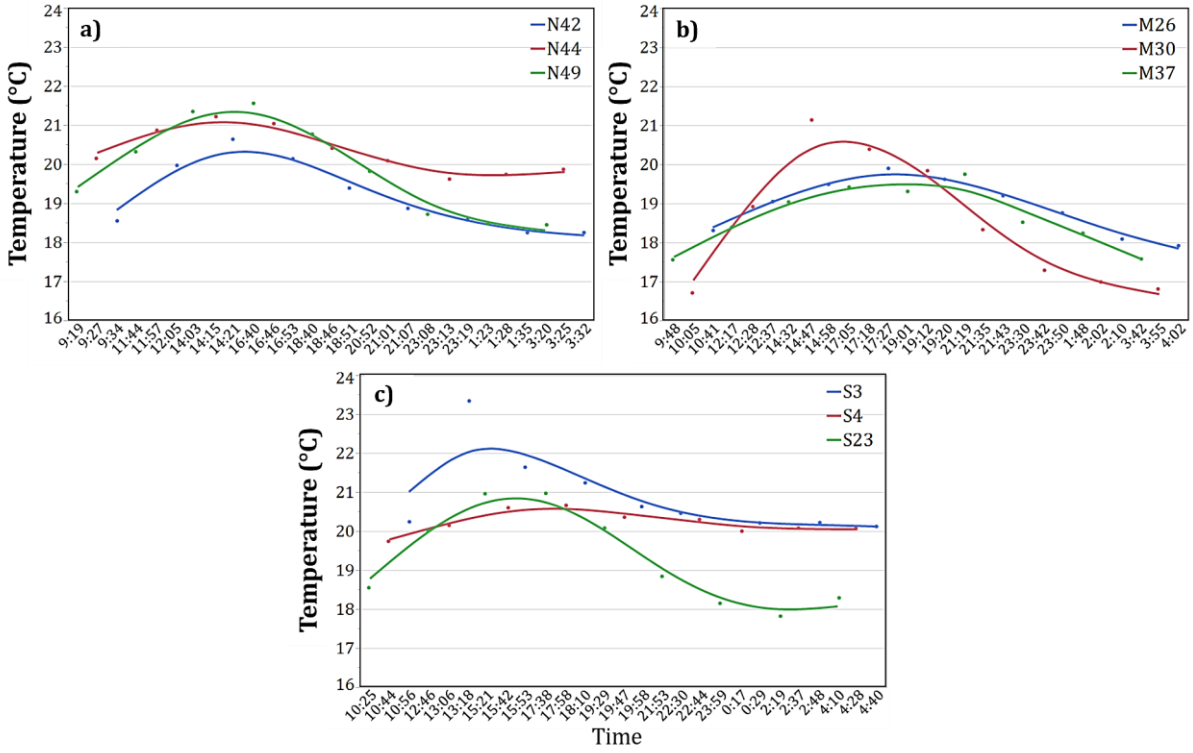

Figure 1.12: Temperature measured over 24-hours periods in three ponds in each of the a) north b) middle c) south areas

The salinity levels in the northern, middle and southern ponds remained virtually constant throughout the 24-hour period, except for a solitary increase in S23 from 40‰ to 50‰, which is likely to be due to inconsistency in the positioning of sampling, considering the constancy of salinity in the overall dataset. Corresponding to the 33 spatial samples (Fig. 1.11b), the northern ponds salinity levels were significantly greater than the middle and southern areas (one-way ANOVA $F_{8,80} = 6.07$, $p < 0.0001$; post-hoc Tukey test, $p < 0.05$), while values in the south and middle ponds remained consistently close to 35-40‰ and were not significantly different ($p > 0.05$; Fig. 1.13a-c).

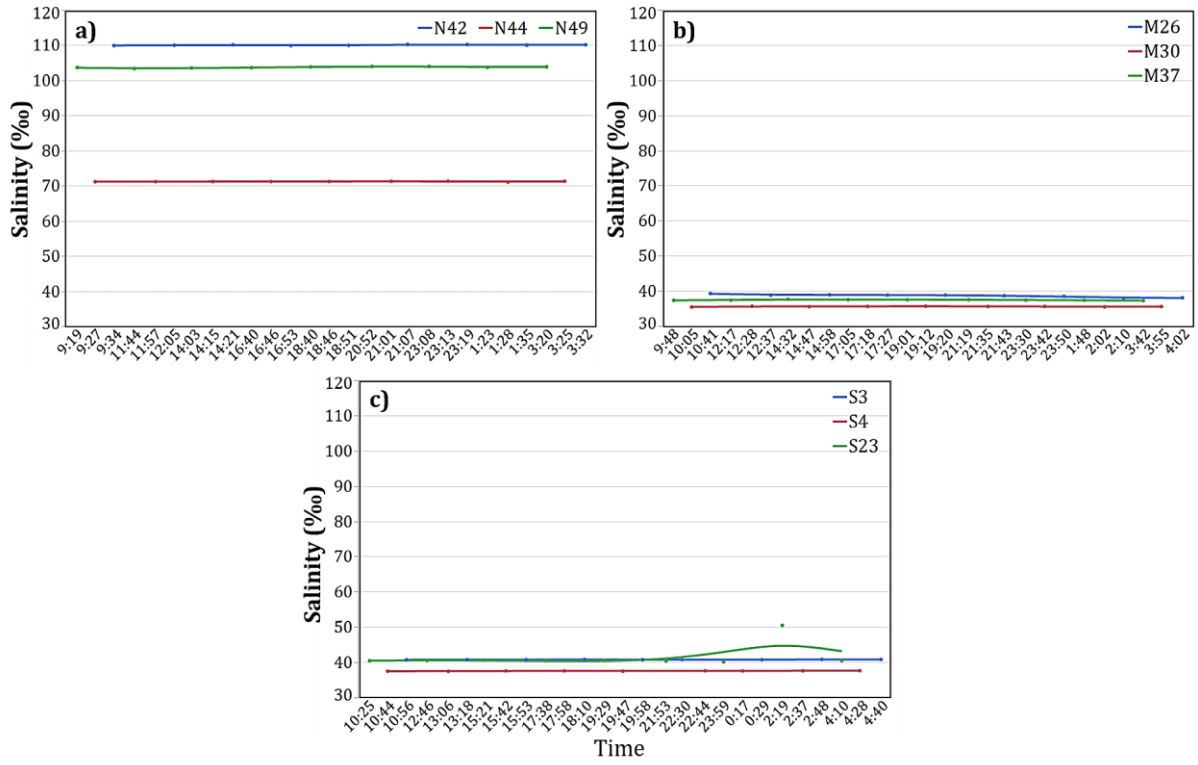

Figure 1.13: Salinity measured over 24-hours in a) north b) middle c) south ponds

Dissolved oxygen displayed the same profiles and trends (Fig. 1.14) as temperature. Significant differences existed in the mean values among areas (one-way ANOVA $F_{8,80}$ = 11.62, $p < 0.0001$), but post-hoc Tukey tests showed that only two ponds (M30 and M37) differed significantly from the means of other ponds ($p < 0.05$), both having greater values. The lowest mean dissolved oxygen recorded was 99%, at S4, and the highest 126% at M30 (Table 1.5). From all individual measures over 24-hours, the lowest dissolved oxygen recorded was 81.1% (S23) and the highest 142.2% (M30) (Fig. 1.14).

In all areas, there was a gradual increase in oxygen concentrations late morning through to noon, reaching the highest levels midday. Beyond this time-period, the oxygen levels steadily dropped late afternoon through the night.

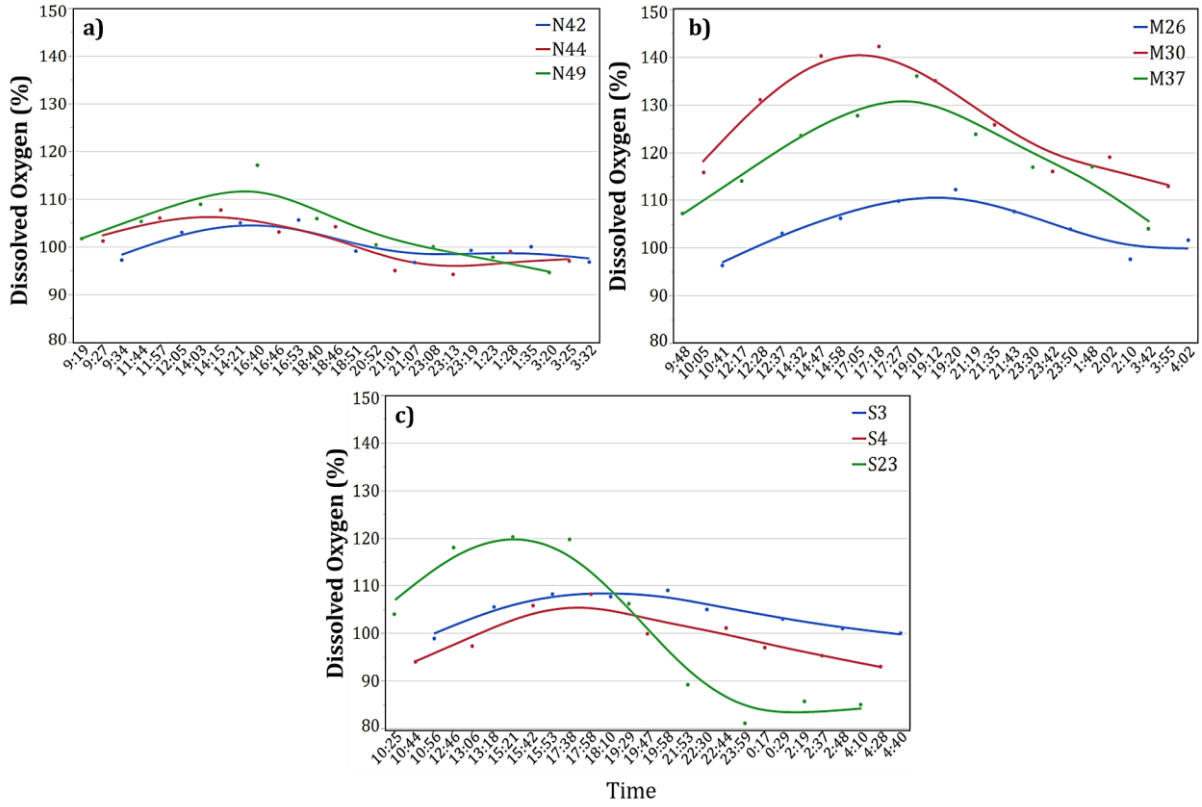

Figure 1.14: Dissolved oxygen measured over 24-hours in a) north b) middle c) south ponds

Although there were significant differences among the mean chlorophyll-*a* levels in the three areas (one-way ANOVA, $F_{8,80} = 77.32$, $p < 0.0001$), post-hoc Tukey tests revealed only one clear difference: that pond M26 stood out as having values significantly greater than all other ponds (Fig. 1.15). This same pond also had a high value when spatial comparisons were made of the ponds (having the second highest value overall; see Table 1.1). It was also the only pond in which a strong diurnal cycle was evident, peaking in the late afternoon.

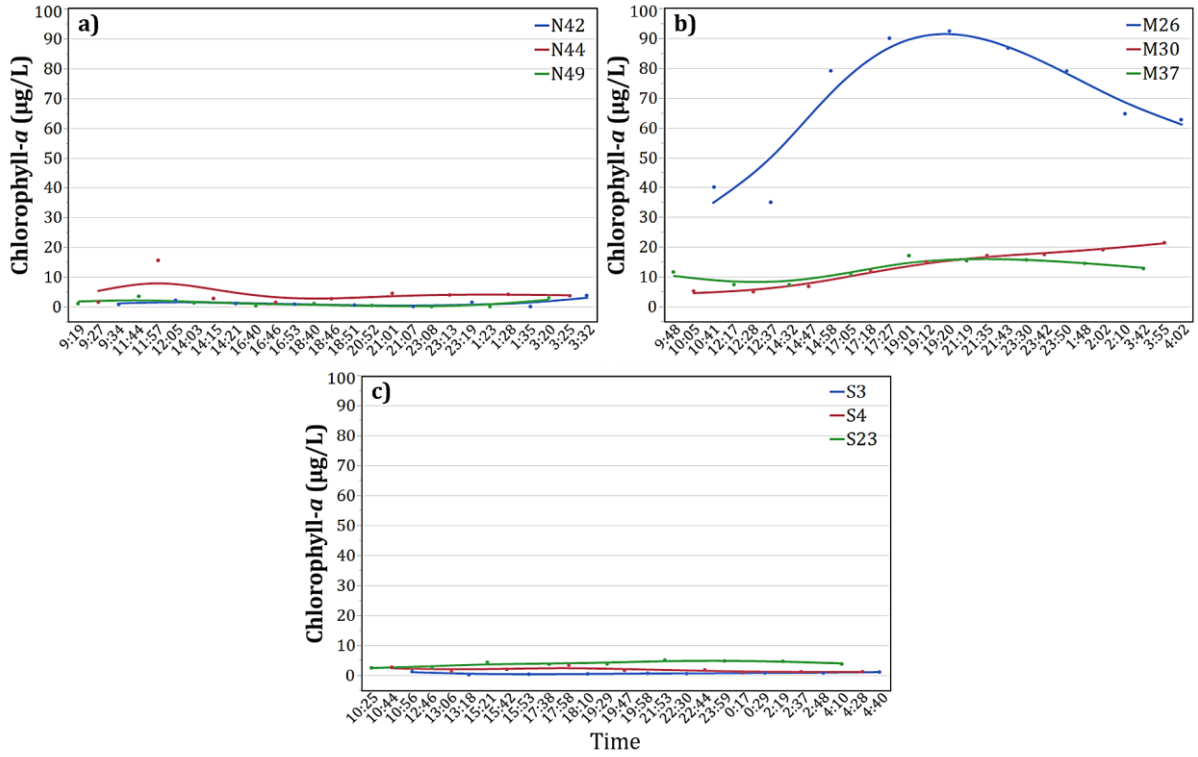

Figure 1.15: Chlorophyll-*a* measured over 24-hours in a) north b) middle c) south ponds

Mean values of pH measurements taken over the 24-hour period differed significantly among ponds (one-way ANOVA $F_{8,80} = 29.11$, $p < 0.0001$). The three northern ponds had the lowest pH values, and post-hoc Tukey tests indicated that the two ponds with the lowest values N42 and N49 were significantly lower than all other ponds, while the third pond (N44) had a value significantly less than most of the remaining ponds ($p < 0.05$). Although the patterns were not always consistent, the pH tended to be higher around midday (Fig. 1.16). From all the samples collected over 24-hours, the lowest pH recorded was 7.67 (N42) and the highest 8.33 (S23).

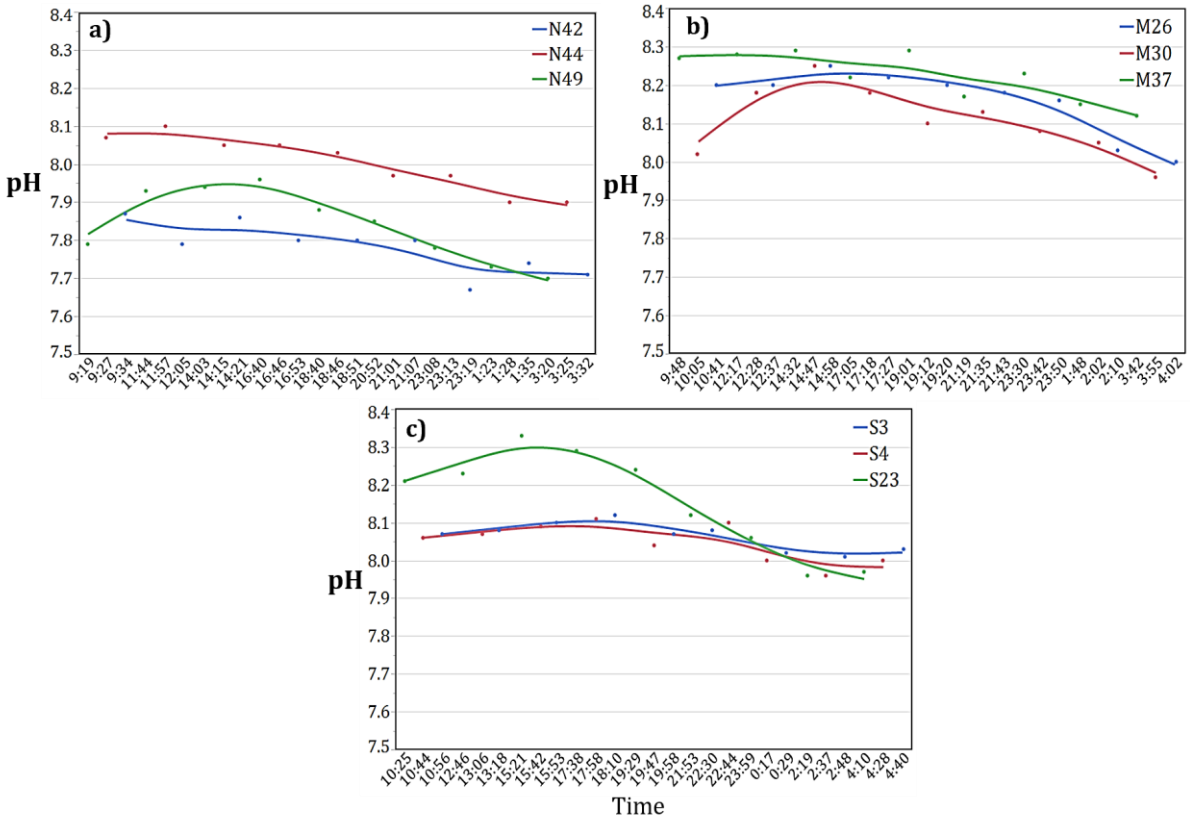

Figure 1.16: pH measured over 24-hours in a) north b) middle c) south ponds

Relationships among variables:

There were significant correlations between dissolved oxygen concentration (%) and pH in six of the nine ponds over the 24-hour monitoring (Table 1.6).

Table 1.6: Correlation between Dissolved Oxygen (%) and pH for ponds (N = 9) measured for 24-hours. Significant values indicated in red

Pond	Variables	Mean	SD	Correlation	Probability
North 42	Dissolved Oxygen	100.29	3.45	0.2835	0.4598
	pH	7.78	0.07		
North 44	Dissolved Oxygen	100.82	4.83	0.7256	0.0269
	pH	8.00	0.07		
North 49	Dissolved Oxygen	103.52	6.71	0.8981	0.0010
	pH	7.84	0.09		
Middle 26	Dissolved Oxygen	104.24	5.31	0.4960	0.1744
	pH	8.16	0.09		
Middle 30	Dissolved Oxygen	126.43	11.14	0.8722	0.0022
	pH	8.11	0.09		
Middle 37	Dissolved Oxygen	118.9	10.03	0.3862	0.3046
	pH	8.22	0.06		
South 3	Dissolved Oxygen	104.26	3.72	0.6679	0.0493
	pH	8.06	0.04		
South 4	Dissolved Oxygen	99.07	5.22	0.7284	0.0260
	pH	8.05	0.05		
South 23	Dissolved Oxygen	101.01	16.08	0.9177	0.0005
	pH	8.16	0.14		

Considering the spatial differences among the 33 ponds, a strong linear relationship emerged between oxygen and pH among the 33 ponds ($R^2 = 0.5267$; Fig. 1.17). There was, however, no significant correlation between oxygen and chlorophyll-*a* ($r = 0.05$, $p > 0.05$).

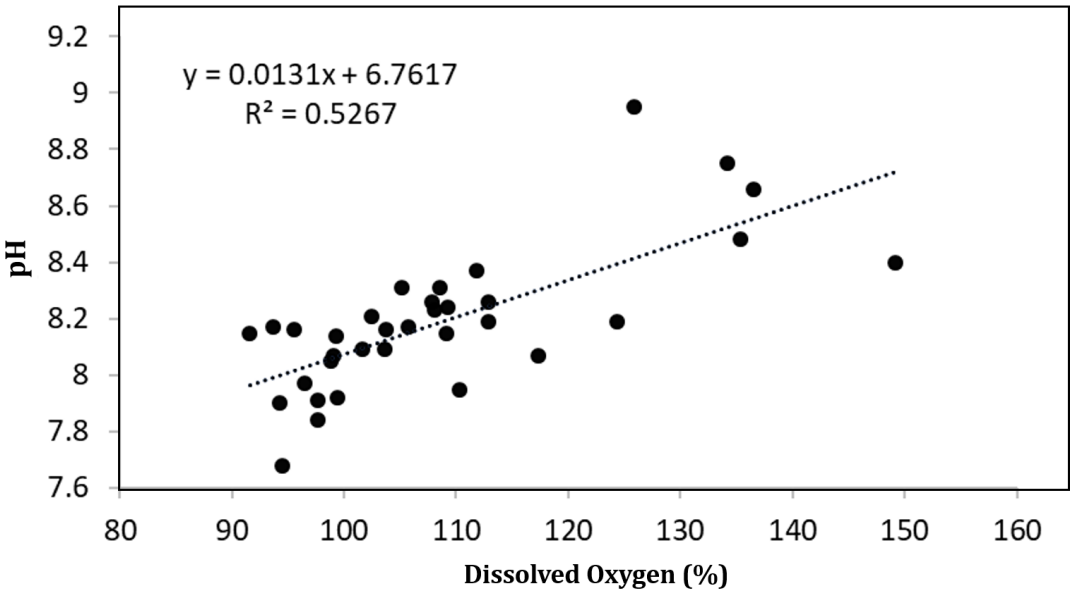

Figure 1.17: Linear relationship between oxygen (% saturation) and pH

Further analysis of the 33 ponds showed that the age of ponds (time since mining) was positively related to salinity in a relationship best fitted by a 3rd-order polynomial regression, for which the coefficient of determination (R^2) was 0.7916 (Fig. 1.18).

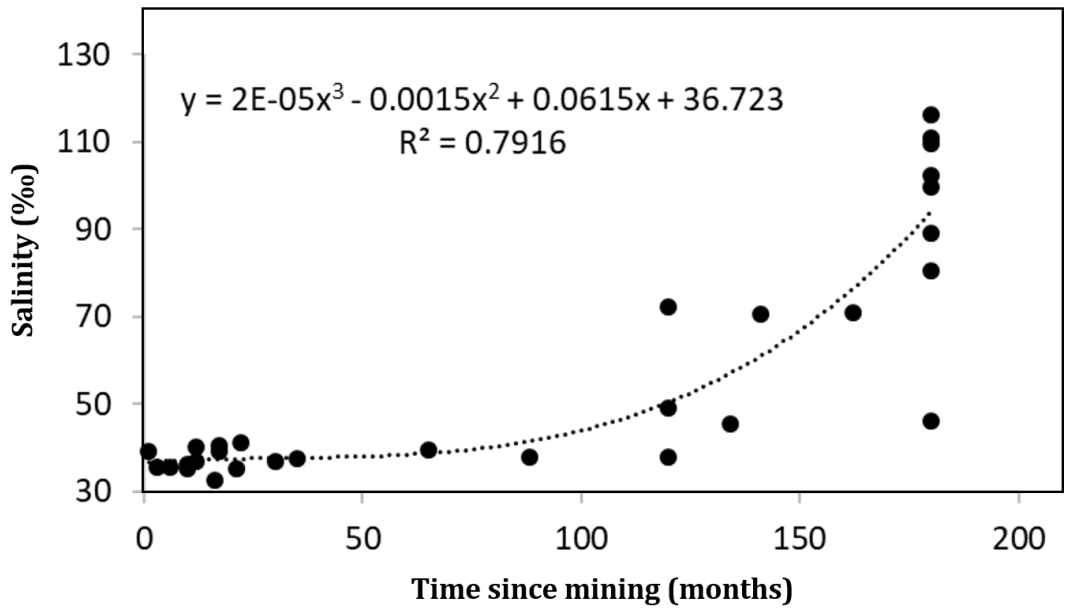

Figure 1.18: Third-order polynomial regression between salinity (‰) and age of ponds (months since mining)

For the same data set, salinity was negatively related to chlorophyll-*a* concentration (Fig. 1.19), with chlorophyll levels being high and spanning a wide range of values at salinities < 50‰ but being consistently low at salinities greater than that. The relationship was best described by an exponential fit (R^2 = 0.397).

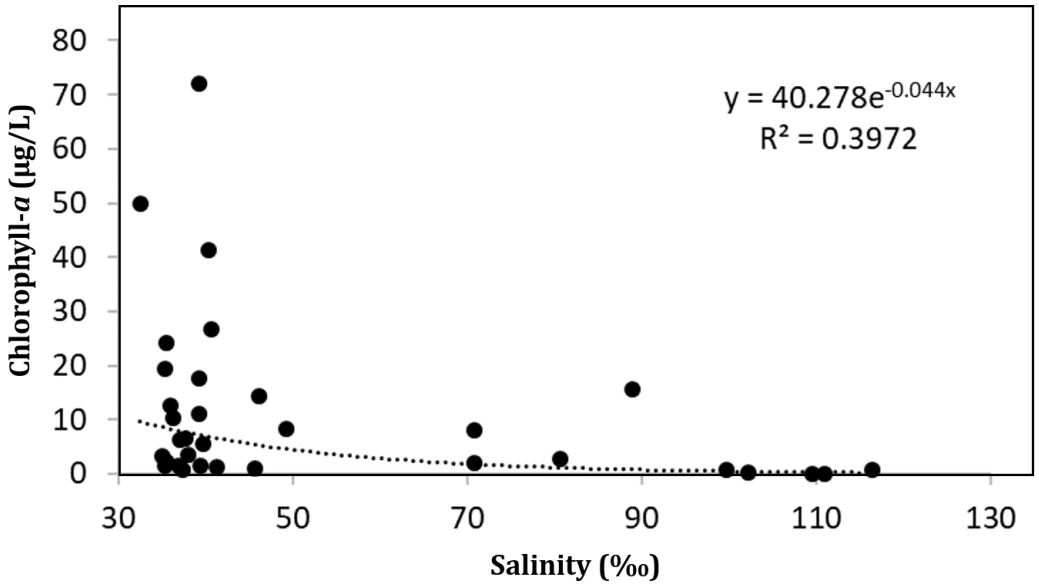

Figure 1.19: Negative exponential relationship between chlorophyll-a and salinity

DISCUSSION

Of critical importance to assessing the ecosystem functioning of the mining ponds is whether physical conditions fall within tolerable limits. Their role and their biodiversity will be determined by the range of physical conditions experienced. Although they may provide habitat analogous with closed estuaries, they are not equivalent because there is no riverine input into them. However, Whitfield (2005) considers Langebaan Lagoon, which lacks riverine input, to be a type of estuary, which her terms a 'coastal embayment'. Moreover, Day (1959) has pointed out that it is not so much salinity that defines estuarine life, but the existence of calm-water conditions, and in this sense the mining ponds fall within the ambit of estuarine conditions. Irrespective of their terminology, the mining ponds clearly constitutes saline coastal wetlands, of which there are only 14 in Namibia (Noli-Peard and Williams 1991). Wetlands on general are recognized as being under

threat, with 58% being classed as 'Critically Endangered' in South Africa (Skowno et al. 2019). Assessing the ecology of the ponds is thus of regional importance.

Several trends emerged from the examination of the physical properties of the ponds. (1) Temperature was higher in the ponds than the sea, higher in the north ponds than middle or south ponds, and increased by day but dropped by night. (2) Salinity was strikingly higher in north ponds than the other ponds or in the sea, but with two exceptions, never departed more than 7‰ from oceanic levels in the middle and south ponds, and in all ponds was constant over the day/night cycle. (3) pH did not differ among ponds, but was more alkaline than in the sea; modest increases in pH were recorded during daylight hours. (4) Oxygen concentrations were not significantly different among the three areas, being consistently high in all ponds; rose by day and fell by night, but never dropped below 80%. (5) Chl-a levels were on a par with, or exceeded those in the sea. Higher mean values were recorded in the middle ponds, with two ponds having values indicative of eutrophication. The trends for each variable are discussed more fully below.

Temperature

The sea-surface temperature along the Lüderitz-Orange River Cone is largely influenced by wind-induced upwelling (southerly and southwesterly winds being predominant) but varies seasonally with temperatures ranging between 11°C and 18°C during summer and 11°C and 14°C during winter (Peard 2007; Hutchings et al. 2009).

The temperature in the mining ponds will depend on local atmospheric temperatures and seasonal trends. My survey was, however, a once-off one with the object of determining spatial patterns among ponds and diurnal variability, rather than quantifying seasonal differences. In these terms, two main outcomes emerged. The first was that temperatures were significantly higher in north ponds than either middle or south ponds. This can be attributed to their being shallower because of evaporation and wind-driven sediment infilling associated with their greater age. The hypothesis I advanced in the Introduction was thus supported – i.e., that significant differences in temperature would exist among the three areas, with higher values in the north. The second trend was that all ponds were substantially warmer than the sea, which is to be expected because solar heating of the

relatively small volumes of water in the ponds will be greater, coupled with the fact that upwelling in the sea lowers temperatures (Shannon 1985; Verheye et al. 2017).

The strong day/night cycles for temperature in all nine ponds are a logical outcome of solar heating by day and cool air temperatures by night. Even the lowest temperatures recorded over the 24-hour period were, however, substantially greater than in the sea.

One of the main effects of water temperature on the ponds' ecological processes is related to oxygen availability in the water, as the amount of oxygen that water can hold decreases as the temperature of the water increases (Whitfield and Bate 2007). In addition, temperature influences rates of photosynthesis and respiration, and the balances between the two alter oxygen levels (Pal and Chakraborty 2014). None of the temperatures recorded are likely to impose limiting stresses on the biota in the ponds. Indeed, they may enhance growth rates to the benefit of oyster mariculture ventures in the ponds, where Pacific Oysters (*Crassostrea gigas*) were successfully farmed during the time of the study. Gagnaire et al. (2006) studied the effects of temperature and salinity on the same species and found that the oyster's haemocytes can adapt to higher temperatures up to 35°C.

While the once-off sampling confirmed that the temperatures of the northern ponds were greater than in the south or middle ponds, it was one of the three southern ponds (S3) that displayed the highest temperatures during the 24-hour monitoring. This pond lies next to S4, which is located approximately 120 meters from the coastline and periodically (during springtides) breaches and overflows, connecting it with S3 (Fig. 1.20). The water in both ponds is pumped out to a nearby Sampling Plant, which treats diamondiferous gravel. In addition to any breaching, the ponds fill up naturally through seepage as they are both located close to the coastline.

During the time of the study, the water level of S3 was lower than usual as no breach had taken place recently, and water was being pumped continually from it to the Sampling Plant, which would have reduced water levels and possibly caused temperature to rise to levels greater than those normally associated with south ponds.

Figure 1.20: Locations of 24-hour monitoring sites S3 and S4. Yellow line shows point of periodic breaching and establishment of connections between ponds S4 and S3. Map derived from Google Earth.

Salinity

On average, salinity in the Benguela Current ranges between 34 and 36‰ (Dingle and Nelson 1993; Robertson et al. 2012). Due to the construction of seawalls, the mining ponds are cut off from the influences of the ocean and consequently salinity levels would be expected to rise in the mining ponds over time, given the low inputs of freshwater and the evaporation anticipated in this arid region. Any changes in the salinity levels within the mining ponds will result in changes in the variety and types of organisms present (Forbes and Cyrus 1993; Whitfield and Bate 2007). Salinity scarcely changed over the 24-hour monitoring. However, a strong long-term trend emerged, with salinity increasing with the age of ponds, unless they were breached to admit an influx of seawater, or were close enough to the sea to experience seepage from the sea. For ponds less than 10 years old, salinity levels in the ponds remained close to those of seawater; but thereafter (notably in the north ponds), values rose steeply to as much as 115‰, exceeding the tolerance levels of most estuarine and marine species (Whitfield et al. 1981). Whitfield (2019) recorded the salinity ranges of various fish species in southern African estuaries and found that less than 20 species have upper limits exceeding 69‰ while over 60 species can survive in water with a salinity level as low as 1‰, making many fish more

tolerant to low rather than high salinity levels. Therefore, any fish species that had entered the older ponds that now have high salinities would have been unlikely to survive when salinities approached or exceed their tolerance ranges.

It was a surprise that the salinity in the ponds remained relatively close to that of seawater for such prolonged periods of time. Likely contributing factors would have been seepage from the sea into ponds, possibly promoted by the fact that the artificial walls would have had a high porosity because of their construction from sediments of mixed origins and particle sizes, coupled with deposition of moisture from coastal fogs, which have been demonstrated to be a more constant source of moisture than local rain, producing a precipitation of around 180 $mm.y^{-1}$, and the reduction of evaporation by the cooling effects of these fogs (Sealy et al. 1998; Henschel and Seely 2008).

Acidity and alkalinity (pH)

The pH is a measure of the acidity or alkalinity of water, which also influences ecological processes (Verheye et al. 2018). The pH in the mining ponds will be influenced by the sources flowing in, namely (limited) fresh water via fog and rain, and seawater via seepage, over-washes and periodic breaching. Seawater pH normally ranges between 7.9 and 8.2 (Whitfield and Bate 2007) and estuarine waters are usually within the range of 7.0 and 8.5 (Snow and Taljaard 2007). Values I recorded in the ponds spanned 7.6 to 8.3, and were thus not beyond those that estuarine organisms will experience and tolerate.

Changes in pH levels in marine systems correlate with changes in temperature, dissolved oxygen and phytoplankton production (Mwegoha et al. 2010). Based on my 24-hour monitoring data, a correlation did exist between dissolved oxygen and pH in the majority of ponds; and a strong linear relationship was detected in the spatial comparisons among ponds. Overall, the pH levels increased throughout the day following the same time trends observed with temperature and dissolved oxygen. When dissolved oxygen increases, carbon dioxide is withdrawn and the mining ponds water becomes less acid resulting in the pH levels increasing. The highest pH values occurred around noon, during periods of high productivity when oxygen peaked and, associated with this, carbon dioxide would have diminished due to photosynthesis, so that changes in temperatures replicated the changes in pH and oxygen (Pal and Chakraborty 2014).

Dissolved oxygen

Variations of dissolved oxygen in the mining ponds are likely to occur seasonally in relation to local atmospheric temperatures and biological activity, and typically peak during daylight hours when plant photosynthesis exceeds respiration, and decrease at night due to respiration (Mwegoha et al. 2010).

During wash-over events, low oxygen bottom water in the ponds can be replaced with fresh oxygenated seawater, depending on the volume of seawater that enters the pond. Wind also plays an important role in distributing oxygen, as proven in a previous study conducted at Dias Point (Lüderitz) where wind-induced changes to the water column were observed in temperature, salinity, oxygen and nutrient chemistry (Peard 2007). The mixing of the mining ponds surface water by wind will increase the rate at which oxygen from the air can be dissolved/absorbed into the water. The depth to which this mixing supplies oxygen to deeper portions in a pond will depend on a pond's size and depth.

My key result was, however, to show that oxygen never descended to level that would be ecologically limiting.

Chlorophyll-*a*

Chlorophyll-*a* is used as a proxy for the abundance of photosynthetic life and can be used as an indicator of the trophic state of ponds (Oxborough 2004; Fondriest Environmental 2017).

Kunlasak et al. (2013) studied the relationship of dissolved oxygen with chlorophyll and phytoplankton composition in Tilapia fishponds where phytoplankton biomass, measured as chlorophyll-*a*, reached eutrophic levels, especially in shallower ponds where temperature increased more quickly. Due to water temperature being an important factor affecting phytoplankton growth (Rhee and Gotham 1981; Richard and Tamar 1987), and light penetration decreasing with depth, the depth differences of the mining ponds in different areas could result in different chlorophyll levels.

Two of the ponds I sampled, M26 and M33, were eutrophic, being visibly green and having high chlorophyll levels exceeding 70µg/L, and two other ponds (S7 and M29) fell between 40-50µg/L; but all other values were equivalent to, or only slightly above, the range that can be expected in the ocean (Chapman and Shannon 1985; Lamont et al. 2019). Because there was no correlation between chlorophyll and oxygen, the daily cycles of oxygen recorded cannot be attributed to changes in biomass of the phytoplankton, but rather to diurnal fluctuations of photosynthetic activity.

Ecological implications

High salinity and chlorophyll levels and diurnal trends in temperature, dissolved oxygen and pH carry implications for species present in the mining ponds.

a) Salinity will rise with the age of the mining ponds unless breaching/seepage introduces adequate seawater supplies. The regression of salinity against age indicates that pond salinities are likely to remain tolerable for about 10 years, but after that, salinity will climb steeply to intolerable levels above 70‰ after about 15 years, causing osmoregulatory stresses in marine organisms, and particularly fish (Whitfield et al. 2006) trapped inside the mining ponds.

   Whitfield (2019) summarises numerous studies in St Lucia Estuary in South Africa regarding biotic responses to increasing salinity and found that fish are more tolerant to low rather than high salinity levels and that very few fish species are able to tolerate salinity values between 70 and 110‰ (see also Whitfield et al. 2006). To survive, fish have to adapt to changes in salinity, particularly during periods when food sources, such as macroinvertebrates that can only tolerate salinity levels up to 60–65‰, die off (Wooldridge et al. 2016), further decreasing the viability of fish (Whitfield et al. 2006; Whitfield 2019). The extent to which fish can survive before salinity levels become lethal is also influenced by temperature (Whitfield et al. 1981).

b) Although temperature was higher in all ponds than in the sea, it never reached levels where fish and macroinvertebrates would not have survived. Whitfield (2019) found that fish mortalities occur during sudden temperature drops when salinities levels are low. At least in subtropical estuaries of southern Africa, even summer

temperatures do not fluctuate greatly and usually range from 24-28°C, which suggests the temperatures recorded in the ponds will not necessarily limit the fish life there (Whitfield 2019).

c) The amount of oxygen in the mining ponds will vary naturally, both seasonally and due to daily cycles influencing the rates of photosynthesis and respiration activity of organisms inside the ponds. Other chemical factors such as temperature, pH and salinity will also affect the dissolved oxygen levels (Mwegoha et al.,2010). The mining ponds were expected to experience anoxic conditions due to the seawalls restricting oceanic supplies of fresh oxygenated seawater to the more saline bottom water in the ponds. However, in reality, oxygen levels never fell below 80%, even during the lowest points of the 24-hour cycle. Similar to micro-estuaries, the high dissolved oxygen levels could be due to the ponds' small sizes and shallow depths, which would have allowed wind-induced water movement to increase the rate at which oxygen from the air will be absorbed into the water (Whitfield 2019).

d) If the pH of the water is too high or too low, fish and other organisms living within the mining ponds will become stressed or die, but it is unlikely that the range of values recorded will limit life in the ponds, particularly as the values were always more alkaline than in the sea and never above 8.4. Although pH decreases the availability of CO_2, Chen and Durbin (1994) found that photosynthesis and growth of marine phytoplankton only start declining at a pH of over 8.8. Abowei (2010) also found that most fish cannot differentiate between pH ranges of 5.5 to 10 and that most productive waters have a pH of 8. Overall, the mining ponds' pH was never above 8.4, always being more alkaline than the sea with no likelihood of acidification issues.

e) Overall, with two exceptions, chlorophyll levels were equivalent to or trivially exceeded those in the sea, which is itself productive, so it is unlikely that there will be a shortage of food supply from this source. In two cases there were indications of eutrophication: Pond M30 had very high levels of chlorophyll and a strong diurnal cycle and Pond M33 had very high levels. The highest level was recorded in M26, which is a shallow pond, connected to the seawater intake channel feeding the main

Treatment Plant in the mine. Water is constantly pumped into the seawater intake channel from surrounding ponds. The channel is shallow, < 5 meters, and the cycle of chlorophyll in M26 tracked irradiance over 24-hours, peaking in the afternoon.

CONCLUSION

Prior to diamond mining in southwest Namibia, the coast was covered by a natural dune hummock system, which has disappeared because of the type of mining conducted in the area, and replaced by an alternative habitat in the form of ponds. The ecological role of these mining ponds in Southern Coastal Mines has been overlooked in the past, mostly because of the difficulty of access and a lack of long-term monitoring data or understanding of the responses of ponds to changes in the mining methodology.

The spatial comparisons of water quality and monitoring over 24-hour cycles revealed that temperature, dissolved oxygen and pH levels had similar diurnal trends but chlorophyll levels barely changed except for one pond, and a positive correlation was found between dissolved oxygen and pH. Salinity levels remained constant during the 24-hour monitoring, but were significantly higher in the north, reaching ecologically stressful levels, as reflected in a negative association between chlorophyll and salinity.

Monitoring water quality in any ecosystem can give valuable insights into the availability of resources that support the ecological systems within a water body (Pal and Chakraborty 2014). From the water quality data collected, it can be concluded that the physical conditions of the mining ponds are such that they can serve as a functional equivalent to natural closed estuaries, but that there is a finite period of about 10-15 years over which they can fulfil this function, after which salinities are likely to rise to critical levels unless contact with the sea is renewed.

The chapters to follow will provide detailed studies of representative mining ponds with a focus on how these ponds serve functional ecosystems in terms of saltmarsh distribution, bird conservation and fish colonisation. The combination of these chapters goes to the heart of the question: are the processes and interactions within the ponds serving a useful ecological role that is worth preserving?

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CHAPTER 2: Saltmarsh Vegetation Associated with the Mining Ponds

INTRODUCTION

Namdeb has the right to practice open-cast diamond mining in south-western Namibia, within an area known as Tsau //Khaeb (Sperrgebiet) National Park. To undertake mining in the marine coastal zone, the shoreline has been accreted offshore and seawalls progressively extended seawards to allow mining beneath sediments that previously lay in the surf zone. As a result of this, ponds have formed in large, deep depressions created by mining in the landward side of the seawalls, into which seawater seeps, forming wetland habitat that is scarce along the coast apart from the Orange River Estuary and effectively generating coastal ponds of considerable size comparable to 'closed estuaries' (see Fig. 1 in Chapter 1).

In this chapter, I focus on the salt marsh vegetation associated with these ponds. The nearest natural saltmarsh vegetation associated with wetlands can be found on the south bank of the Orange River floodplain (Fig. 2.1) and, more than 250 kilometres further north, in the bays of Lüderitz, which support limited intertidal wetlands and saltmarsh vegetation, covering less than 5 km^2 (Matiza and Chabwela 1992).

Brown and Day (1959) surveyed the Orange River estuary as part of their research on the ecology of South African estuaries. They identified the dominant saltmarsh species at the time as (1) *Arthrocnemum* (now *Salicornia*) *perennis* (closest to the water) – which may have been misidentified and in reality is likely to have been *Salicornia natalensis*, and (2) *Arthrocnemum africanum* (further from the water) – now known as *Salicornia pillansii*.

Bornman et al. (2004) later investigated the adaptation of the saltmarshes along the Orange River. Their study confirmed that the dominant saltmarsh species at that time was *Salicornia pillansii* (recorded under the name *Sarcocornia pillansii*). They described how this saltmarsh plant is able to access underground water sources but that in many parts of the estuary, the groundwater becomes too saline to be used by the plant. The hypersaline groundwater also affects the soil salinity, which, combined with the low rainfall rates in the area, reduces seedling recruitment and natural recolonization of *S.*

pillansii. This has impacted the health of the saltmarsh directly, which threatens the Ramsar status of the Orange River mouth.

Bornman and Adams (2010) outlined how the Orange River Ramsar site was placed on the Montreux Record in 1995 (a record of Ramsar sites in which changes in ecological character have occurred, are occurring or are likely to occur), due to the poor state of the saltmarshes on the southern bank. The largest anthropogenic impact on the Orange River estuary marshes was a causeway built in the 1960s on the South African side (adjacent to Alexander Bay), which separated the saltmarshes from the main river channel and hindered water from flowing into the area, degrading the marshes (Shaw 2007; Shaw et al. 2008). A natural flood in 1988 inundated the area east of the causeway, flooding the saltmarshes, and the causeway then prevented water from draining out again, resulting in further die-off of the marshes (Shaw 2007).

These events degraded about 27% (approximately 3 km^2) of the salt marsh. Bornman and Adams (2010) and Shaw (2007) investigated whether the remaining salt marsh vegetation would produce sufficient seeds to revegetate the area. Both studies found that the depth of the water table and the increase in groundwater salinity (which caused the sediment electrical conductivity to exceed the tolerance limit for *Salicornia pillansii*) still had the greatest influence on the loss of this plant along much of the Orange River.

In this chapter I provide an overview of the diversity and amount of salt marsh vegetation growing around the periphery of the mining ponds, to determine if the ponds provide significant amounts of suitable habitat for salt marshes, in view of the fact that the ponds lie on a coast where this habitat is largely absent because of its arid nature. Specifically, the chapter will consider whether the ponds can act as stepping stones for the redistribution between widely dispersed salt marsh vegetation, which would add to their conservation value.

My analyses address a series of hypotheses:

1. My primary hypothesis was that the ponds will support saltmarshes that would otherwise not exist in the region, providing habitat for them between the far-strung saltmarshes in the Orange River Estuary and the lagoons of Lüderitz. More specifically, I addressed the following secondary issues and hypotheses:
2. I quantified the amount of saltmarsh vegetation associated with the ponds, to assess the importance of the ponds relative to the natural saltmarshes previously documented at the Orange River Estuary. I hypothesised that the proportion of ponds with saltmarshes and the amount of saltmarsh growth would increase northwards because of the greater length of time the ponds there have been in existence.
3. I also examined the diversity of saltmarsh species associated with ponds, and hypothesised that the diversity would be less around the ponds than at the Orange River Estuary, where the range of habitat types will be greater due to tidal action and a gradient of conditions up the estuary.
4. I determined whether the health and prevalence of the saltmarsh vegetation differed between the northern, middle and southern ponds, with the expectation that there would be increasing proportions of healthy plants from north to south areas, given that the age of ponds (and their salinity and isolation from the sea) increases northwards.
5. I evaluated whether the positioning of the saltmarshes around the periphery of the ponds was indicative of their being introduced by wind transport. I hypothesised that the saltmarshes would grow mainly on the north-easterly ends of the ponds due to the wind being directed predominantly towards the north and north-east (Peard 2007).

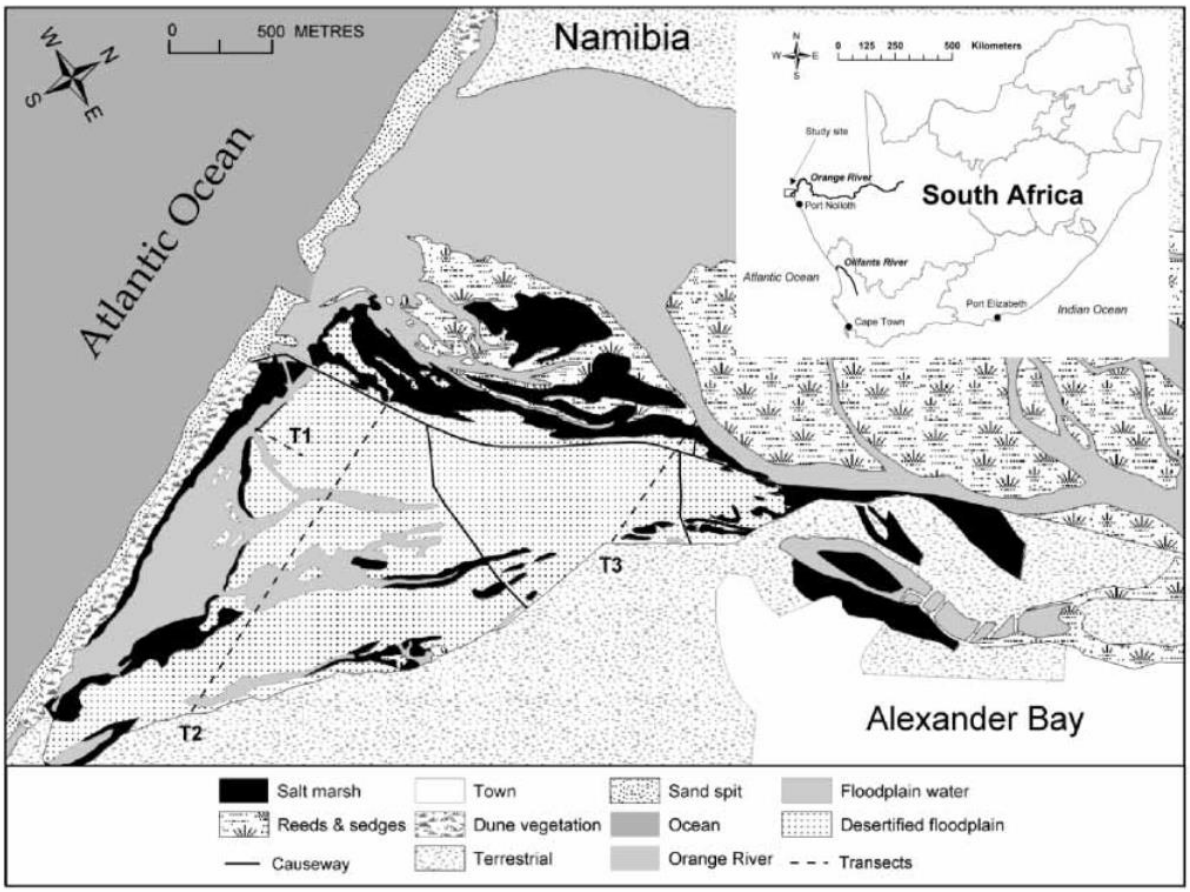

Figure 21: Salt marsh vegetation along the Orange River wetlands (after Bornman et al. 2004)

MATERIALS AND METHODS

Study area

The sites I examined lie within the Southern Coastal Mines, Namdeb's southernmost mining license area, which extends approximately one hundred kilometres north of the Orange River mouth, the longest river in southern Africa and one of few perennial rivers on the arid south-west African coast (Simmons and Allan 2002).

The distance between the most southern and most northern mining ponds is approximately 75 kilometres, with 150 ponds in total stretching along the coast. The study area was divided into three sections namely north, middle and south, according to the age and status of mining activities within each section. The northern area, which contains older ponds where mining ceased more than 15 years ago, is approximately 50km in length and contained 111 ponds. The middle area contains 15 younger ponds where active mining has occurred recently or is still occurring, and is approximately 10km in length. The southern ponds, where 'dewatering' (removal of water) and mining

are current or recent, is approximately 15 km in length and contains 24 ponds. Sampling was carried out at all accessible mining ponds during summer in December 2017.

Based on visual evidence, saltmarsh vegetation was present in all three areas, but appeared more established along the northern ponds' boundaries. As the ponds age and change in size, the saltmarshes change in nature, with healthy green growths occurring closest to the water's edge, stressed red plants above them, and dead areas highest above the shores. As a null hypothesis, proportions of healthy, stressed and dead growth were postulated to be equal among the northern, middle and southern areas, but it was anticipated that this null hypothesis would not be upheld, and my alternative hypothesis was that the relative amounts of healthy saltmarsh would be greater than expected in the south rather than in the north.

Sampling method

The study area was divided into three sections namely north, middle and south, based on differences in the post-mining ages of the ponds and their proximity to the sea (as outlined in Chapter 1).

At each accessible site where saltmarsh vegetation was present, photographs and samples from flowering saltmarsh plants were taken for identification. The total length of each saltmarsh was measured and five randomly chosen transects were run across the saltmarsh from top to bottom to measure the widths of saltmarsh occupied by healthy (green), stressed (red) and dead (shrivelled and brown) plants (Fig. 2.2). The pond perimeters were measured using Google Earth (Fig. 2.3).

To relate the orientation of saltmarshes to wind direction, I plotted rose diagrams of the frequency with which the centres of saltmarshes fell in each of sixteen 22.5° quadrants, and compared that with a wind rose derived from Meteoblue.com showing the number of days/year that wind blew in the direction of each quadrant for nearby Oranjemund.

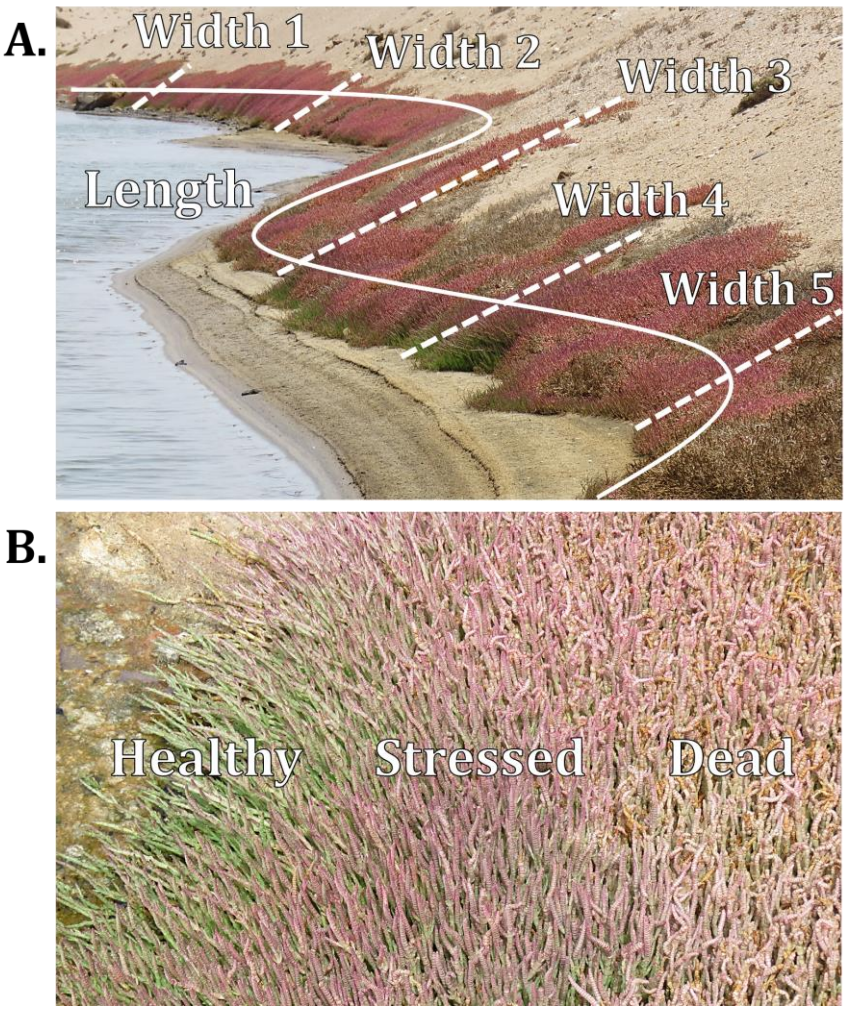

Figure 2.2: Field sampling method a) measurement of total length and five transects, b) health category classification

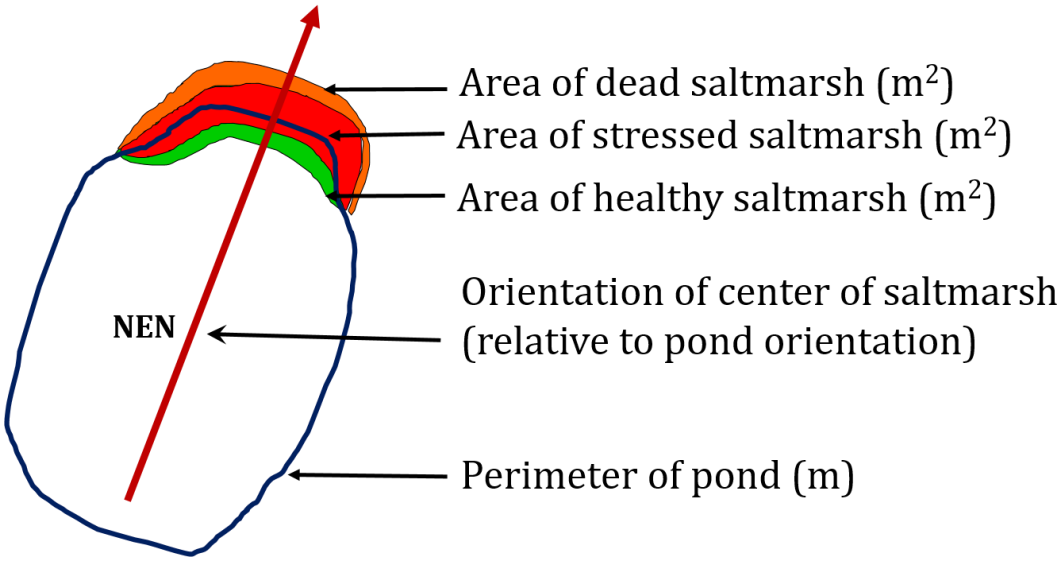

Figure 2.3: Calculations of total areas of Healthy/Stressed/Dead saltmarshes (m^2). Red arrow indicates the centre of orientation of the saltmarsh on each pond relative to sixteen 22.5° compass quadrants

Statistical analysis

Data were analysed using statistical software JMP (version 14) which is a suite of computer programs for statistical analysis developed by the Statistical Analysis System Institute (SAS).

Measurements from the total length and five transects at each pond were used to determine the overall saltmarshes condition per area. The total saltmarsh cover per pond was converted to amounts of healthy, stressed and dead saltmarsh by multiplying the total length of saltmarsh per pond by the average widths of healthy, stressed and dead growth (determined from the five transects per saltmarsh). The data were then analysed with one-way ANOVA tests to see if the relative amounts of healthy, stressed and dead salt marsh (in m^2) differed among the three areas. Pilot Generalised Linear Models demonstrated that pond perimeter had no significant effect on the areas of saltmarsh, so it was dropped from the analyses.

To relate the orientation of saltmarshes to wind direction, I derived a Pearson correlation coefficient between the percentages of saltmarshes whose centres lay in each of sixteen 22.5° quadrants and the average number of days/year that the wind blew in the direction of each quadrant.

RESULTS

Cover of saltmarshes

The area in the north had the largest saltmarsh cover of 605231 m^2; the middle had a covering of 79927 m^2 and the south had 98128 m^2 (see Fig. 2.4). Total measured area of the pond-associated saltmarshes was thus 0.78 km^2. Allowing for the fact that 42% of the ponds could not be sampled (see below), the total amount present would have been around 1.34 km^2.

Of the 150 mining ponds, 24 were located in the south, 15 in the middle and 111 in the north. From the 111 northern ponds, 85 marshes were measured around 41 ponds. Six ponds had no growth (Fig. 2.5). The remaining 64 ponds in the north were unsafe to access due to the lack of maintenance on the gravel roads after mining ended in the area more than 15 years ago. All middle and southern sites were accessible and 11 saltmarshes

were measured from 9 of the 15 middle ponds. The remaining 6 middle ponds had no saltmarsh growth. Lastly, 14 saltmarshes were measured from 8 of the 24 southern ponds and the remaining 16 ponds had no saltmarsh growth.

The proportions of ponds that supported saltmarshes declined significantly from North, through Middle to South ponds (Fig. 2.5, Table 2.1; Chi-squared = 21.48; df = 2, $p < 0.001$). The null hypothesis of no differences in the frequencies of 'growth/no growth' among the regions North, Middle and South was thus rejected and the alternative hypothesis that there were significant differences in the amounts of growth and no growth in the North, Middle and South was accepted, with growth being greater in the north than the middle or south.

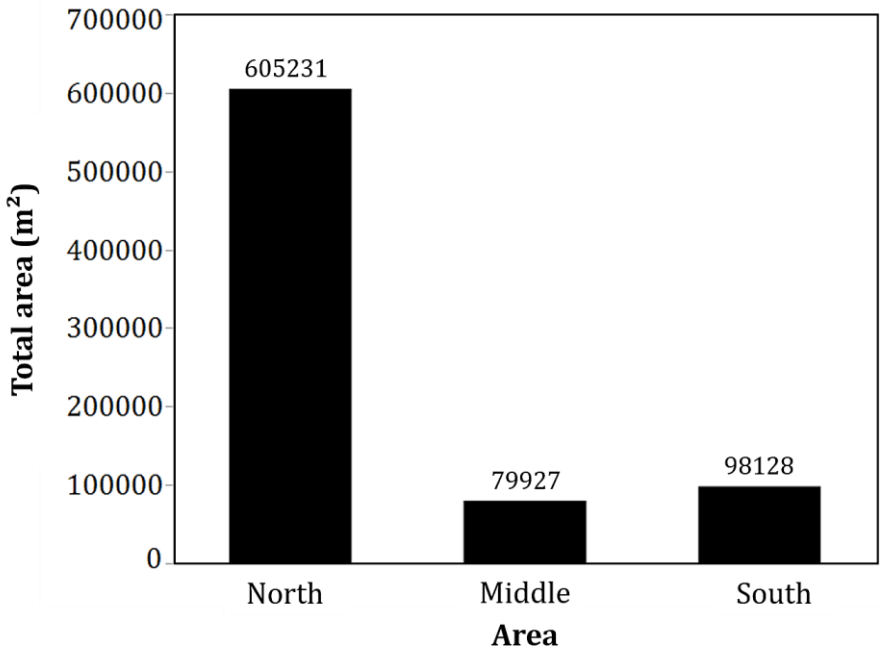

Figure 2.4: Total saltmarsh cover per area (m^2)

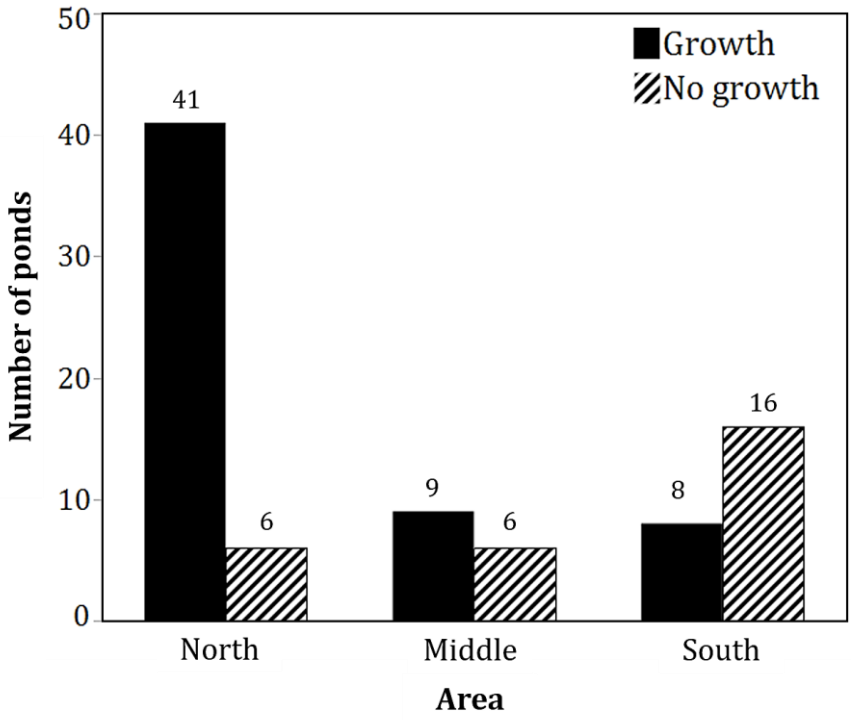

Figure 2.5: Saltmarsh growth around ponds per area

Species

The vegetation samples collected were all from one species namely, *Salicornia natalensis affinis*, a sprawling, prostrate shrub with a knobby structure and diagnostic flowers that are concealed by the bracts of each segment, with only the stamens protruding (Fig. 2.6).

Figure 2.6: *Salicornia natalensis*, showing how flowers are concealed by pink ring-like bracts, with only white anthers protruding

Saltmarsh conditions

Healthy saltmarsh growths were found at the edges of ponds where the soil moisture content is favourable (Veldkornet et al. 2015). As soon as ponds begin to shrink due to evaporation or dewatering, the healthy green growth follows the water downward, leaving the more established saltmarsh further away from the water to become stressed and eventually dying.

In total, 110 marshes were measured from the northern, middle and southern areas. The healthiest saltmarshes grew in the south (19.51%) followed by the middle (8.30%) and then north (4.72%). More than half of the southern (51.14%), middle (60.96%) and northern (64.88%) saltmarshes were stressed and nearly equal proportions of dead growth were found in the south (29.35%), middle (30.74%) and north (30.40%) (Fig.

2.7). One-way ANOVAs revealed that the differences in the amounts of healthy saltmarsh among the three areas were significant ($F_{2,107} = 14.37$, $p < 0.0001$), whereas those for stressed and dead saltmarsh were not (respectively $F_{2,107} = 1.426$, $p = 0.2447$, and $F_{2,107} = 0.72$, $p = 0.4915$).

Saltmarshes were not well established around the middle ponds due to active mining and excavating activities in the area. The overburden material stripped from mining sites is used to maintain the seawalls and mining cross-walls that protect the mines from flooding. After an area is mined out to bedrock level, it naturally begins to fill with seawater through seepage. With the overburden removed, there is limited sediment available for saltmarshes to become established in the mined-out areas, except along the seawalls and mining cross-walls (Fig. 2.8).

In the south, the majority of the marshes grew around dewatered ponds, on the seawall side (southwest) where seepage occurred (Fig. 2.9). Saltmarshes began to colonise such areas within 6 months, and substantial beds formed within 16 months (Fig. 2.10). However, the marshes are destroyed as soon as mining starts in an area and excavators remove overburden material including any saltmarsh growth.

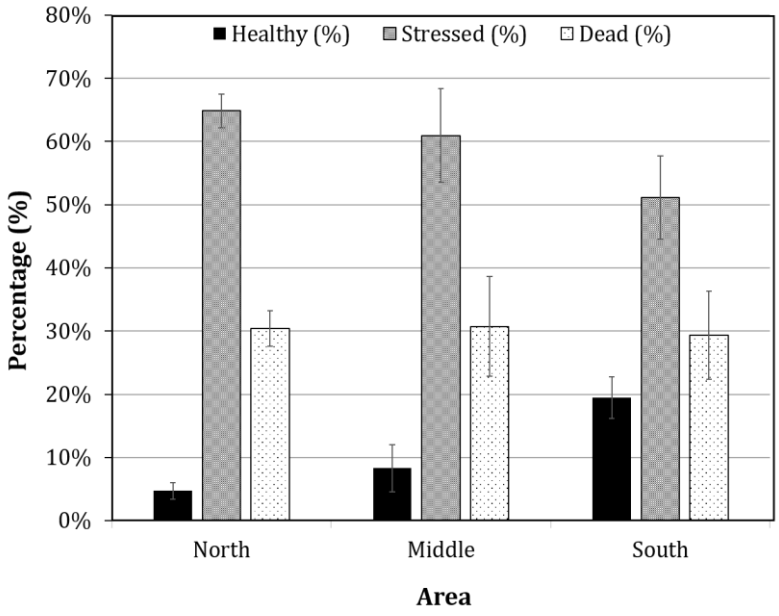

Figure 2.7: Healthy, stressed and dead growth per area

Figure 2.8: Recently mined out middle pond filling up naturally through seepage from the seawall

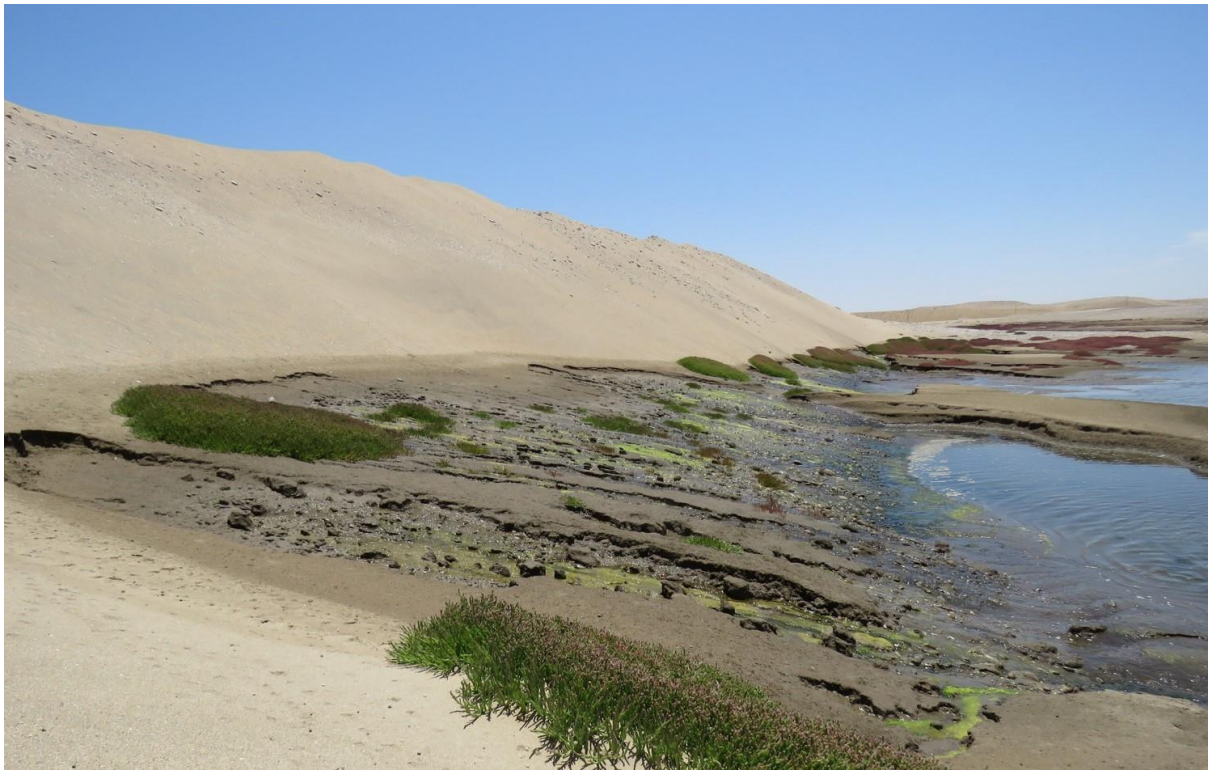

Figure 2.9: Saltmarsh growth near seepage inlets along a southern pond edge

Figure 2.10: Establishment of saltmarsh growth over 16 months in a dewatered southern pond

Table 2.1: Chi-square results for differences in the amounts of 'growth' and 'no-growth' among north, middle and south areas

Category	Observed (O)	Expected (E)	O-E	$(O - E)^2$	$(O - E)^2/E$
Growth	41	31.69	9.30	86.53	2.73
No Growth	6	15.30	-9.30	86.53	5.65
Growth	9	10.11	-1.11	1.25	0.12
No Growth	6	4.88	1.11	1.25	0.26
Growth	8	16.18	-8.18	67.01	4.14
No Growth	16	7.81	8.18	67.01	8.58
Total Chi-square					21.48

NOTE: df = 2, Criteria 9.210, Chi stat > Chi Criteria, $p < 0.0001$

Position of growth

From the 110 marshes measured, the highest number (31 or 28.2%), were oriented with their centers of distribution on the NNE side of the mining ponds. The second highest orientation was SSW (26 or 23.6%), and the third highest NNW (24 or 21.8%).

In the north area, the three highest growth orientations were NNE (28.2%), SSW (27.0%) and NNW (20.0%). The middle ponds' highest growth positions were NNW (36.4%) and NNE (27%). Lastly, the southern ponds highest growth positions were NNE (28.6%) and both NNW and NE (21.4% each).

Considering all three areas collectively, rose diagrams reflecting the centers of orientation of saltmarshes lay prevalently in either a NNW to NNE direction or a SSW direction (Fig. 2.11a). Winds blew prevalently towards N, NNW, NNE and SSW directions (Fig. 2.11b), and the orientation of saltmarshes was correlated significantly ($r = 0.659$; $df = 14$; $p < 0.005$) with direction towards which winds blew (Fig. 2.12).

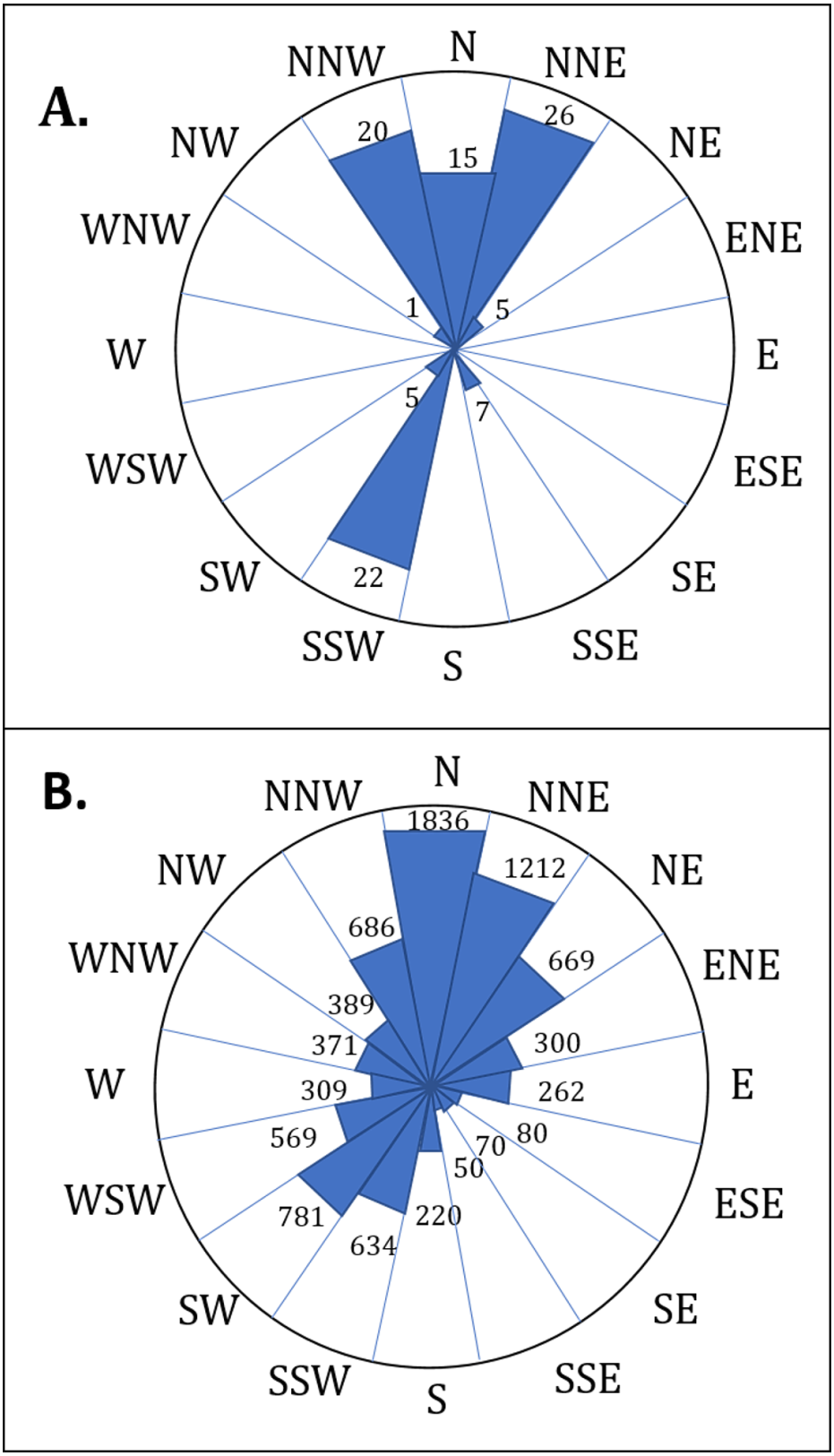

Fig. 2.11: Orientation of saltmarsh growth relative to wind direction. a) Percentages of saltmarshes with their center of orientation falling in each 22.5° quadrant. b) Direction towards which wind blows (number of hours per year per 22.5° quadrant)

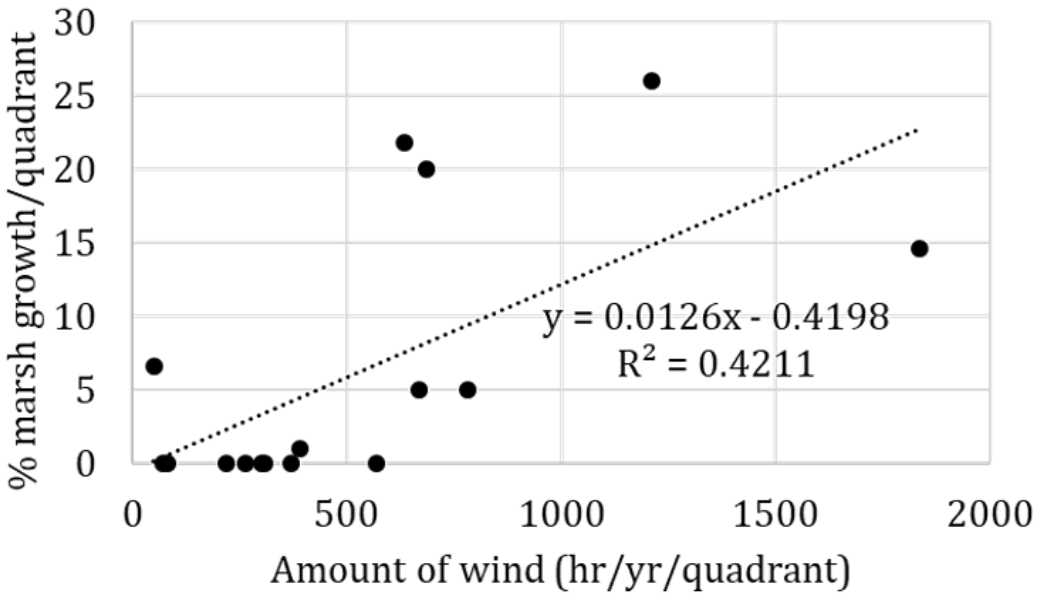

Figure 2.12: Saltmarsh growth near seepage outlets along southern pond edges. (Coefficient of determination, $R^2 = 0.422$; correlation, $r = 0.659$; $n = 16$, $p < 0.005$.)

DISCUSSION

My first key finding was that although saltmarshes readily became established around most ponds, they comprised a single species only, namely *Salicornia natalensis affinis*. My first hypothesis of expecting the ponds to support saltmarshes that would otherwise not exist in the region is thus supported, but seems applicable to just one species. *Salicornia natalensis affinis* is often found as solitary plants or beds around the edges of saline wetlands of South African estuaries, mainly near the coast (O'Callaghan 1992). In the mining area, the proportion of ponds with saltmarsh increased northwards, supporting the second of my hypotheses that because of the greater age of ponds in the north, there would be greater amounts of saltmarsh there.

The saltmarshes associated with the mouth of the Orange River Estuary are, by contrast, much more diverse, with no less than 11 species of coastal saltmarsh species (Veldkornet and Adams 2013), despite degradation of the saltmarshes by reductions of water flow, causeway construction and flooding (Shaw et al. 2008). One on the most common species at the Orange River Estuary, *Salicornia pillansii*, which is usually found in the supratidal saltmarsh zones of South African estuaries and is a dominant species in areas with high sediment electrical conductivity and high groundwater salinity (Veldkornet and Adams 2013; O'Callaghan 1992), did occur in the mining region I examined, but was never

associated with the ponds, occurring on dunes landwards of the ponds. The third of my hypotheses, that the saltmarshes around ponds would be impoverished relative to the Orange River Estuary, was thus also upheld. No alien vegetation species were recorded along the mining ponds, unlike the situation at the Orange River where several alien genera (*Nicotiana, Prosopis, Eucalyptus* etc.) have established themselves due to the presence of two nearby towns, and seeds being washed down the river (Simmons and Allan 2002).

The estimated total area of the pond-associated saltmarshes (1.34 km^2) amounted to about 26.8% of that present as natural saltmarsh at the Orange River Mouth (Matiza and Chabwela 1992; Shaw 2007). For *Salicornia natalensis* specifically, the proportional area it occupies around ponds would have been much greater than this, as the species occurs only around the littoral fringe of the saltmarshes on the Orange River Estuary.

The mining activities in each area, namely south (with current dewatering and active mining), middle (active and recently mined areas) and north (historical mining but no recent mining activities) have had an important influence on how saltmarshes form, survive and degrade among the ponds. From the surveys, three things were evident. (1) The saltmarsh growth was healthier in the south where water seeped through the seawalls and mining cross walls. This supported my fourth hypothesis. The marshes quickly became established nearby these seepages although they were destroyed by mining activities that removed sediment. (2) The middle area had the least growth due to active mining activities. (3) The northern area had no current mining activities, allowing marshes to become established and maintained around the ponds. However, the marshes there were stressed to a greater degree than elsewhere, which could be attributed to their higher salinity levels (Chapter 1) which may cause the sediment electrical conductivity to exceed the tolerance limit for *S. natalensis*.

Soil salinity is an important factor influencing the growth and survival of saltmarshes, and the related species *S. pillansii* has an upper salinity tolerance of about 80 $mS.cm^{-1}$ (Shaw 2007), equivalent to approximately 51.8‰, the units in which I measured salinity in Chapter 1. Soil salinity was not measured during my study, but my measurements of water salinity showed that the salinity levels rise with the age of the mining ponds, with

the northern ponds water salinity levels being more than double those in the middle and southern ponds (Chapter 1). In the south, the constant seepage of seawater through the seawall and mining cross walls allows the salinity to remain within limits tolerable by the saltmarshes. Coastal fog is also an independent source of moisture for saltmarshes (Sealy et al. 1998; Henschel and Seely 2008).

However, another factor is that water levels in the ponds decline over time due to evaporation, leaving the upper portions of the saltmarshes desiccated so that they become stressed and eventually die, while at their lower edges propagation of fresh runners extends the plants down the shore.

Wind plays an important role in the seed dispersal of saltmarsh plants (Shaw 2007). The mining ponds lie parallel to the Atlantic Ocean within the Benguela System, which is dominated by southwesterly winds, and is one of the strongest coastal upwelling centres in the world (Andrews and Hutchings 1980; Peard 2007). Such strong winds can disperse large amounts of seeds or even plant material across long distances. The significant correlation I obtained between wind direction and the orientation of saltmarshes around the ponds does support the idea that wind plays an important role in seed dispersal and the positions at which saltmarshes become established, and upholds the fifth of the hypotheses I advanced in the Introduction. Neff and Baldwin (2005) considered that water dispersal was considerably more important than wind dispersal for the distribution of saltmarsh plants; but they were dealing with a tidal system in which water movements would have been substantial. The ponds I examined were all non-tidal, so water transport would have been limited.

The mining ponds lie north of the Orange River mouth, which is recognised as both an Important Bird Area and is one of 17 Ramsar Sites in South Africa (Bornman et al. 2004). The ponds themselves generate a unique habitat for waterbirds (see Chapter 3), which can disperse seeds along the coast in their guts, on their feathers or in mud on their feet (Vivian-Smith and Stiles 1994; Neff and Baldwin 2005).

A number of studies have been conducted since Darwin (1859) first recognised that seeds could be dispersed by various metazoan groups, such as migratory birds, which can

externally transport seeds on their feet and feathers (epizoochory). However, Green and Figuerola (2005) pointed out that Darwin overlooked the possibility of internal seed transport (endozoochory) through the digestive tracts of birds. This not only allows the birds to disperse a variety of seeds over long distances but also to preserve connectivity among aquatic communities that are not coupled hydrologically, as is the case for mining ponds (Reynolds and Cumming 2016; van Leeuwen et al. 2012).

Various transport vectors influence the frequency and size of seeds transported (Green and Figuerola 2005; Nathan et al. 2008). Six transport mechanisms have been described by Nathan et al. (2008), and my study area exhibits the potential for all six because it possesses the necessary features to promote long-distance dispersal of seeds along the coast. These include: (1) Open terrestrial landscapes: the study area is located in the Succulent Karoo, which is characterised by sparse, low-growing succulents and shrub vegetation with few grasses, and a lack of tall trees and dense vegetation. This open landscape area provides favourable conditions for wind-dispersal of seeds. (2) Animals: saltmarshes play an important ecological role that supports primary production, creates habitats for fish, birds and other wildlife, and stabilises soil around estuaries and shorelines (Shaw 2007). This was evident during sampling where birds were observed collecting dead pieces of *Salicornia natalensis* and pieces of other plants to build nests (Fig. 2.13) and Black-backed Jackals (*Canis mesomelas*) and Brown Hyenas (*Hyaena brunnea*) used the marshes for shelter, resting and feeding points (Fig. 2.14). While feeding on top of saltmarshes, jackals and/or hyenas may incidentally consume or collect seeds on their fur, which according to Nathan et al. (2008) has the potential to disperse seeds further than wind-dispersed seeds due to these animals' large body and gut sizes. (3) Migratory animals: in the absence of trees found in the study area, birds find shelter in the saltmarshes during the day while feeding at the edges of ponds. Birds, particularly migratory ones, are effective transport vectors of seeds due to their frequent flights between aquatic ecosystems (van Leeuwen et al. 2012) and their ability to overcome barriers to seed dispersal (Nathan et al. 2008). (4) Extreme meteorological events: the largest portions of pond edges occupied by saltmarshes lay to the northeast due to prevailing southwesterly winds that can disperse seeds across long distances from pond to pond. Saltmarshes also grew on the southwesterly and southeasterly portions of ponds in the lee of windbreaks from the seawall and mining cross walls, reflecting periodic wind

reversals and eddies. (5) Ocean currents: seeds can float with the northward-moving surface currents of the Benguela. (6) Mining activities redistribute sediments when overburden materials containing seeds are shifted from the mining sites and are deposited on the beach during active accretion of the shoreline.

The saltmarshes growing along the mining ponds boundaries could be acting as a seed source for the area between the Orange River estuary and the lagoons of Lüderitz. My survey was a once-off one with the object of exploring the role of the ponds as stepping stones between largely dispersed estuarine and lagoonal systems along the coast. The majority of the northern area marshes were under stress and their survival depends on abiotic factors such as sediment salinity, moisture content, and groundwater depth (Veldkornet et al. 2015; Bornman et al. 2008). Further studies are required to investigate these abiotic stressors and the survival of seedlings, which is fundamental in the life cycle of marshes (Shaw 2007). My analyses did reveal a correlation between wind direction and positioning of saltmarshes, but a range of other means of dispersal exist, including epizoochory and endozoochory by waterbirds and mammals, and my data do not allow an assessment of their relative importance. Saltmarshes in southern Africa are under threat from a range of environmental changes and in South African about 43% have been lost, diminishing the ecosystem services they provide (Adams 2020). The creation of additional saltmarshes in the mining ponds is thus significant.

Figure 2.13: Different bird species found at ponds: a) Whitebreasted Cormorants nesting b) African Spoonbills building nests c) Cape Teal feeding next to marshes

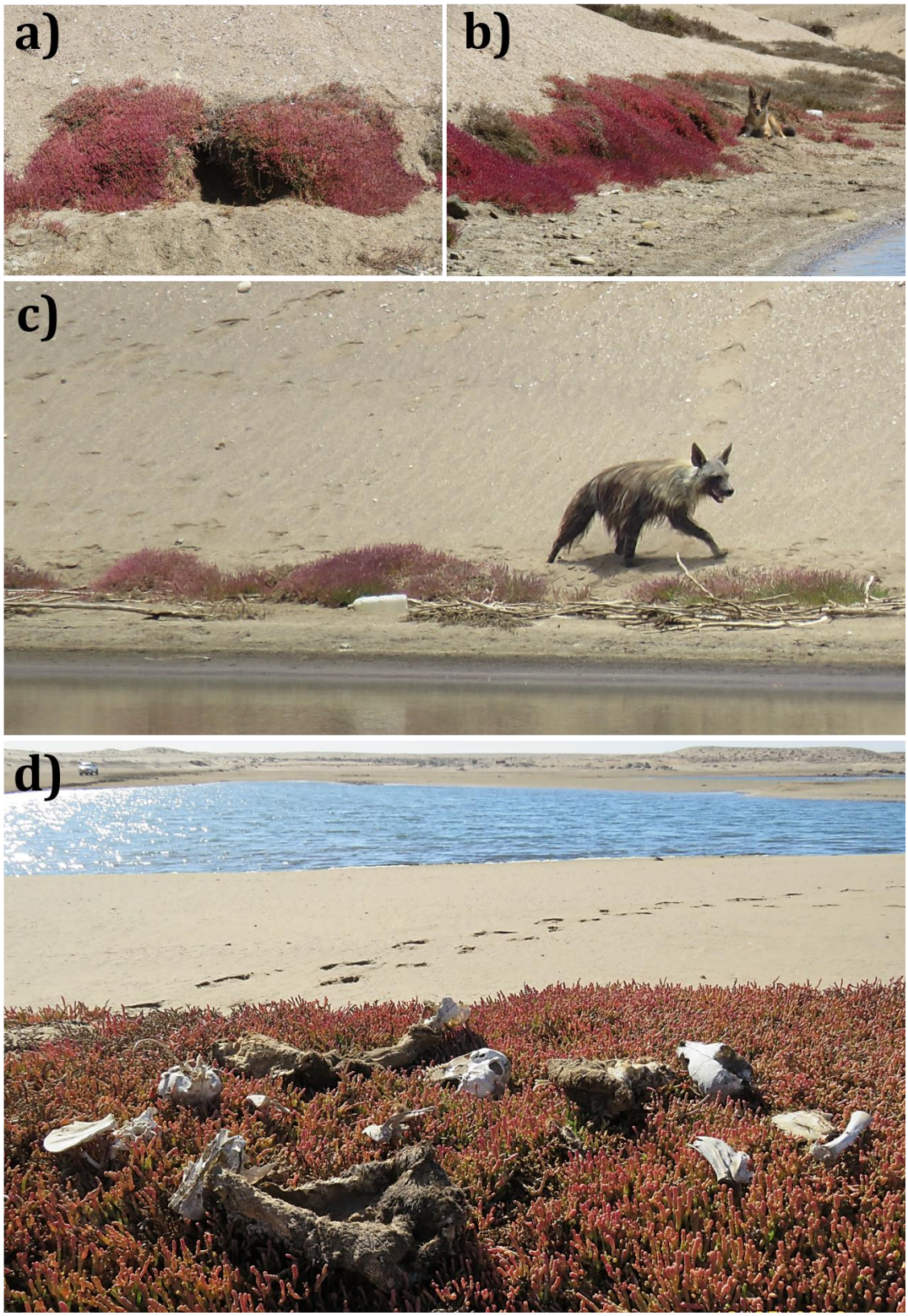

Figure 2.14: Animal activity around saltmarshes a) animal den between marshes b) jackal resting on top of marshes c) Brown Hyena scavenging around pond d) Hyena feeding area on top of marshes

CONCLUSION

Mining operations along the south-western Namibian coastline have resulted in the creation of multiple inland seawater ponds lying immediately adjacent to the coastline. Supporting my first hypothesis (1), the mining ponds do offer an environment for the establishment, growth and distribution of saltmarshes in areas that would otherwise not support this habitat in the region. In terms of my other hypotheses, (2) the proportion of ponds with saltmarshes was greatest in the north, as expected from their greater age; but (3) it was striking that only a single species, *Salicornia natalensis*, seems to benefit from the presence of the ponds, in contrast to the much greater diversity of species in the wetlands of the Orange River Estuary. Nevertheless, the ponds add approximately 28% to the availability of habitat suitable for saltmarshes in the region and, in all likelihood, a much greater percentage of the amounts of habitat suitable for *S. natalensis* specifically. (4) The health of the saltmarshes was greatest in the south, where the ponds are relatively young and their salinity levels approximate those of seawater due to seepage across the seawalls. (5) In keeping with the fourth of my secondary hypotheses, there was a strong correlation between wind direction and the orientation of saltmarshes around the periphery of ponds. Although this is suggestive of wind dispersal of seeds, there are other avenues of dispersal that may contribute to the spread of saltmarshes from pond to pond.

The potential for the pond-associated saltmarshes to permanently establish along the coastline will depend on (a) whether these communities will be disturbed by future mining activities, (b) the progressive increase in salinities as the ponds age, and (c) the rapidity with which ponds will be filled by aeolian sand transport. One of the biggest limitations in understanding the ecological role of these mining ponds is due to the limited number of studies conducted in the area because of access restrictions. The chapters that follow will focus on how the ponds attract birds and support fish populations.

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CHAPTER 3: Birds Associated with the Mining Ponds

INTRODUCTION

Birds are an integral part of wetland ecosystems, which are the most threatened of South Africa's ecosystems, with up to 58% classified as 'Critically Endangered' and only 8% considered 'well protected' (Taylor and Peacock 2018; Skowno et al. 2019). In this chapter, I focus on the birdlife associated with the ponds generated by coastal diamond mining in southern Namibia. The study area is located in the southern-most mining licence area permitted to Namdeb Diamond Corporation (Pty) Ltd, stretching along the coast just north of the town of Oranjemund, close to the border with South Africa, in the restricted diamond area in south-western Namibia. This area falls within the Tsau //Khaeb (Sperrgebiet) National Park, north of the Orange River mouth (Simmons et al. 1999; Anderson et al. 2003). The Namibian Islands' Marine Protected Area is also situated along this southern coastline of Namibia (Currie et al. 2008) and supports large populations of seabirds including several globally and locally endangered species (Kemper et al. 2007; Ludynia et al. 2012).

Despite its arid nature, Namibia supports a large number and considerable diversity of birds, with 644 species having been reported, of which over 90 are endemic to the southern African sub-region and 13 endemic to Namibia specifically (Robertson et al. 1998; Simmons et al. 1998a). The Orange River mouth in particular is the closest wetland to the mining ponds that supports large numbers of waterbirds, with maxima of 20 653 to 26 653 birds of 57 species being recorded within its area of approximately 2298 hectares (22.98 km^2) (Anderson et al. 2003). Because of this, it is recognised as both an Important Bird Area and one of 17 Ramsar Sites in South Africa (Bornman et al. 2004).

Since the Orange River mouth was designated a Ramsar site in 1991, the number of waterbirds has decreased in the area, motivating Anderson et al. (2003) to conduct a study to re-assess the Orange River's Ramsar status. During 20 surveys conducted between January 1980 and December 2001, they found that in spite of the drop in the numbers of birds present, species richness of waterbirds remained relatively constant, with an average of 52 species being recorded.

They concluded the site still meets three of the four Ramsar criteria under which it was originally designated in 1991, namely, (a) it is an example of a rare and unusual wetland type on the arid west coast of southern Africa; (b) it supports 'an appreciable assemblage of rare and endangered bird species'; (c) it supports 'more than 1% of the populations of three endemic species' and 'more than 1% of southern African populations of six species of waterbirds'. The criterion not met is that the site no longer regularly supports in excess of 20 000 waterbirds, primarily as a result of the decline in the numbers of Cape Cormorant *Phalacrocorax capensis* and Common Tern *Sterna hirundo* (Anderson et al. 2003).

Mining operations along the south-western Namibian coastline have resulted in the creation of multiple coastal seawater ponds that generate a unique habitat for waterbirds that may serve as an alternative foraging and breeding site and act as a resting point for migrant birds in an otherwise arid environment. Prior to mining, water is removed from the ponds ('dewatering') and the benthos in the ponds is left exposed. Active mining then follows, during which time disturbance is at high levels. After mining is complete, the ponds are left to refill by seepage. The time elapsed after mining is completed is very different in the north, exceeding 15 y, versus the south and middle ponds, where mining is either current or was completed only recently.

This chapter provides an overview of how many and what bird species are currently benefiting from the existence of these mining ponds, whether there are any seasonal patterns in their abundance, which of these species are of conservation concern, and the numbers of nests, juveniles and carcasses recorded. The data collected in the restricted mining area will be compared with bird counts recorded by other researchers at the Orange River mouth and other wetland areas on the west coast.

My general hypothesis is that the ponds will support and provide habitat for a significant number of waterbirds relative to those in other wetlands in the region.

More specifically, I addressed the following issues:

1. I quantified the species composition and abundance associated with the ponds and the conservation status of these species.
2. I determined whether the species composition differed among the northern, middle and southern ponds with the expectation that there will be greatest numbers and diversity of birds per pond in ponds that are being dewatered, making benthic food available; lowest numbers where mining is current and disturbance is high, and that in the post-mining phase there will be greater numbers in the north ponds, than the middle or south ponds, which are younger.
3. I assessed whether the numbers of birds and the numbers of bird species were related to the age of ponds (years since mining ceased); and if the numbers of birds were related to the salinity of the ponds.
4. Lastly, I evaluated whether there are seasonal patterns among the most abundant birds species.

MATERIALS AND METHODS

Study area

The study site is bordered by the Orange River mouth in the south, the longest river in southern Africa and one of few perennial rivers on the arid south-west African coast (Simmons and Allan 2002; Anderson et al. 2003), and lies adjacent to the nutrient-rich Atlantic Ocean (Simmons et al. 1999).

The distance between the most southern and most northern mining pond is approximately 75 kilometres, with 150 ponds in total stretching along the coast. The study area was divided into three sections namely north (N), middle (M) and south (S), based on differences in the post-mining ages of the ponds (as outlined in Chapter 1).

Sampling method

Eight bird counts (two per season) were conducted at each pond, at roughly equal time intervals from July 2017 to June 2018. Seasons were defined as winter (June to July), spring (September and November), summer (December to January) and Autumn (March and May). See Appendix 3.1 (pp 113-116) for a list of ponds surveyed, their location, size and last-mined date, status of mining, total numbers of birds counted per pond and numbers of species recorded per pond.

The eight counts were conducted during (1) S: 27 July 2017, M: 28 July 2017, N: 31 July 2017; (2) S: 4 Sep 2017, M: 6 Sep 2017, N: 8 Sep 2017; (3) S: 8 Nov 2017, M: 13 Nov 2017, N: 16 Nov 2017; (4) S: 5 Dec 2017, M: 6 Dec 2017, N: 7 Dec 2017; (5) S: 3 Jan 2018, M: 4 Jan 2018, N: 5 Jan 2018; (6) S: 2 Mar 2018, M: 2 Mar 2018, N: 3 Mar 2018; (7) S: 7 May 2018, M: 8 May 2018, N: 9 May 2018; (8) S: 5 June 2018, M: 6 June 2018, N: 7 June 2018.

Birds were visually monitored through binoculars from a distance of at least 50 m to avoid disturbance. The surveys quantified the species, numbers and noted activities (feeding, nesting, presence of chicks/juveniles). Wood's (1946) classification was used to divide young birds into chicks (nestlings/fledglings) and juveniles (fledged/immature young).

No birds were caught: photographs were taken (with a Canon Powershot SX60HS camera) to confirm identifications if any were in doubt. Sinclair et al., (2011) was used for identification, with scientific names being updated when necessary. As the distinction between Common Terns (*Sterna hirundo*) and Arctic Terns (*S. paradisaea*) was difficult at a distance, the two have been combined under the former name, as that species seemed more prevalent. Invertebrates were identified from Branch et al., (2016). No quantitative surveys of invertebrates were undertaken as that was the goal of a separate programme forming part of another thesis (by J. Cloete); but observations were made on the benthic species present to allow me to comment on the availability of food for birds, and to relate the benthos to fish diets in the following chapter.

Similar to the saltmarsh survey (Chapter 2), a majority of the northern ponds were unsafe to access due to the lack of maintenance of roads since mining ended in the area over 15 years ago. Since bird counts were conducted visually from a distance, some of the inaccessible ponds could still be monitored for bird activity. Of the 150 mining ponds, 111 were in the North, of which 55 were monitored; but all 15 middle and all 24 southern ponds were accessible and were monitored. In estimating the total number of birds present at all ponds combined, counts were scaled up to allow for the fact that only 55 of the 111 north ponds were monitored.

Assessment of the importance of ponds

Comparisons of pond-use by birds with their occupation of other wetland systems were based on information in the literature, drawing largely on a series of surveys of Namibian coastal wetlands (cited in Appendix 3.3, pp 118-120). It was not possible to sample closed estuaries equivalent to the ponds, as there are no comparable systems in the region. The most important comparison I could make was with data for the Orange River Estuary, which has been extensively studied (Williams 1986; Simmons and Allan 2002; Anderson et al. 2003). The next nearest substantial wetlands from the Orange River Estuary are approximately 400km to the south, namely Olifants River Mouth, and 500km to the north, at Sandwich Harbour, near Walvis Bay (Kolberg 2002). The closest analogue to the ponds is Cape Cross Lagoon, which was previously an embayment with a connection to the sea, but has been cut off by a sand barrier and now receives seawater only by seepage and washover, which is balanced against evaporative desiccation (Simmons et al. 1999).

To assess the importance of the ponds for bird populations, I compared their wetland bird numbers and species richness with values derived from the literature or websites, as follows: (a) Estimates of population sizes of species in southern Africa, based on Harrison et al. (1997), Hockey et al. (2005), Whittington et al. (2016) for Kelp Gulls, Brooke et al. (1999) for Greyheaded Gull, Tree (1997) for Pied Avocet, Crawford et al. (2013) for White-breasted Cormorant; the Second Southern African Bird Atlas Project (Brooks and Ryan 2019), accessed from SABAP2.adu.org.za; or a link supplied in 'Atlas of Namibia' (Mendelsohn et al. 2002) to stored data accessed from the-eis.com 'Fig. 4.35 Coastal birds – source data.html'; (b) numbers of birds listed in the Coordinated Waterbird Counts (CWAC) specifically for the Orange River mouth (accessed from http://cwac.adu.org.za/); (c) thresholds set in CWAC for species in terms of Important Bird Areas (IBAs), Subregional IBAs, Ramsar sites, or 1% of the regional population; or for1% of the flyway population (Wearne and Underhill 2005).

Statistical analysis

Data were analysed using the statistical software JMP (version 14), developed by the Statistical Analysis System Institute (SAS), to analyse species abundance per area, compared to mining activities and seasonal patterns (nested graphs). One-way analysis of variance (ANOVA) with a significance level of 0.05 was used to determine significant

differences in bird counts among seasons. When significant differences among means were found, post-hoc Tukey honestly significant difference (HSD) tests were run to determine the significance of differences among individual seasons. I also used one-way ANOVA to compare differences of bird counts among the phases of mining.

A second statistical program PAST (Paleontological Statistics) version 2.17c was used to analyse species composition per pond in each area. PERMANOVA was used to determine the significance of differences among areas, cluster analysis to display patterns of similarity among ponds, and SIMPER to determine species similarity and dissimilarity among ponds. Abundances of each species were pooled for the whole year and data were not transformed for statistical analysis for PERMANOVA, Cluster and Dendrogram analysis as they met the assumption of equality of variance. Diversity was explored by comparing numbers of species (species richness), the Shannon Index of alpha diversity and the Shannon evenness index for each of the three areas (north, middle, south). Comparisons of total bird counts per pond versus the areas of ponds (m^2) revealed no significant relationship ($R^2 = 0.0133$, $p > 0.05$), so the areas of ponds were discounted as an explanatory variable. To explore the potential relationship between salinity in ponds (derived from Chapter 1) and numbers of birds per pond, I ran a Pearson correlation analysis. The relationship between age of ponds and either total bird numbers or number of species was explored by regression analysis and Pearson correlation coefficients.

RESULTS

Species abundance

The total count of birds across all surveyed ponds (summed for the eight surveys) was 52408 birds and 36 species (Table 3.1). The north area had the most ponds and the highest bird count of 35181; the middle area had the lowest count, 4166, and the south 13061 (Fig. 3.1a). Extrapolating to allow for the fact that counts were done in only 55 out of the 111 north ponds, total numbers in the north would have been 71000, bringing overall total numbers up to 88228 for the eight counts, or 11028 birds per count. Expressed as numbers per pond, this amounted to averages of 639.6, 277.7 and 544.2 birds per pond in the north, middle and south respectively (Fig. 3.1b). Of the 36 taxa recorded, ten species or subspecies (asterisked in Table 3.1) are regional endemics

(Hockey et al. 2005; Kemper et al. 2007; Sinclair et al. 2011; Simmons et al. 2015). Five of the species are endemic to southern Africa: the African Black Oystercatcher (*Haematopus moquini*), Bank Cormorant (*Phalacrocorax neglectus*), Cape Cormorant (*Phalacrocorax capensis*), South African Shelduck (*Tadorna cana*) and Hartlaub's Gull (*Chroicocephalus hartlaubii*), as are three subspecies: the Blacknecked Grebe (*Podiceps nigricollis gurneyi*), Swift Tern (*Sterna bergii bergii*) and Kelp Gull (*Larus dominicanus vetula*). In addition, the Whitebreasted Cormorant (*Phalacrocorax lucidus*) is restricted to sub-Saharan Africa, and the Greyheaded Gull *Chroicocephalus cirrocephalus poiocephalus*) is endemic to large parts of sub-Saharan Africa and Madagascar.

The International and Namibian conservation statuses for the 36 species found in the ponds are listed in Table 3.1. Of note are the Bank Cormorant, rated as endangered (numbering 149 summed over the eight counts); the Blacknecked Grebe, considered near-threatened in Namibia (18428); Cape Cormorant, endangered (6787); Greater Flamingo, vulnerable in Namibia (5205); Lesser Flamingo (near threatened or vulnerable (1572).

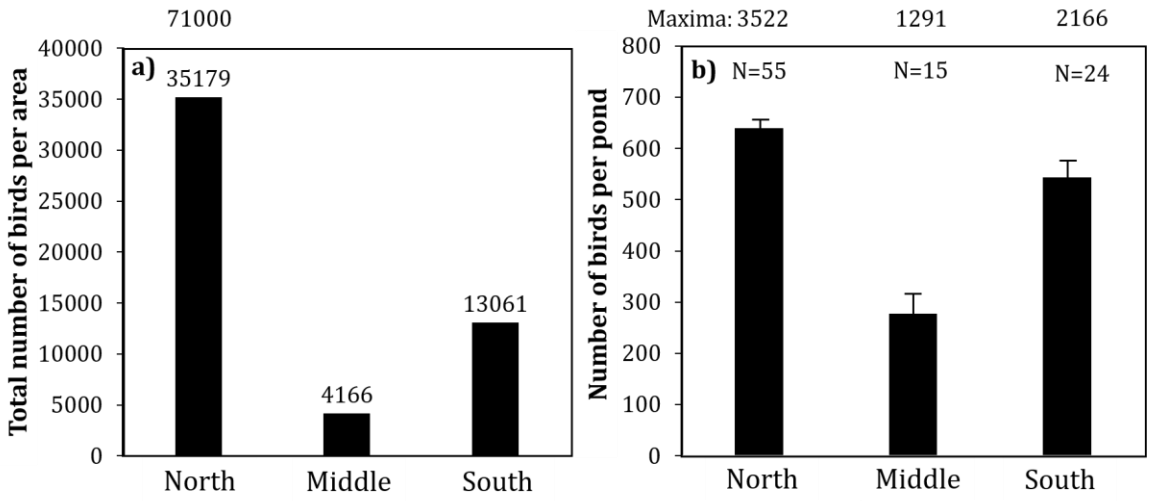


Figure 3.1: a) Total numbers of birds counted in the north, middle and south ponds. The figure of 71000 for the north is the extrapolated estimate for all north ponds. b) Mean (+1SE) numbers of birds per pond (summed over 8 counts) in the north, middle and south ponds. N = number of ponds counted; maximum values for individual counts are listed at the top

Table 3.1: Species classification (ordered on common names), status (Simmons et al. 2015; IUCN 2019), and abundance per area (summed over eight counts during the study period) for 55 north, 15 middle, and 24 south ponds. * indicates species or subspecies regionally endemic.

Species	Scientific Name	IUCN Red List Category	Namibian Status:	North	Middle	South	Total
African Black Duck	*Anas sparsa*	Least Concern	Least Concern	0	3	2	5
African Black Oystercatcher	*Haematopus moquini*	Least Concern*	Near-threatened	0	0	87	87
African Spoonbill	*Platalea alba*	Least Concern	Least Concern	0	0	149	149
Bank Cormorant	*Phalacrocorax neglectus*	Endangered*	Endangered	69	2	54	125
Blacknecked Grebe	*Podiceps nigricollis*	Least Concern*	Near-threatened	18113	134	178	18425
Blacksmith Plover	*Vanellus armatus*	Least Concern	Least Concern	0	0	7	7
Blackwinged Stilt	*Himantopus himantopus*	Least Concern	Least Concern	299	24	46	369
Cape Cormorant	*Phalacrocorax capensis*	Endangered*	Endangered	1089	168	5530	6787
Cape Teal	*Anas capensis*	Least Concern	Least Concern	1369	166	143	1678
Common Greenshank	*Tringa nebularia*	Least Concern	Least Concern	8	1	3	12
Common Tern	*Sterna hirundo* [1]	Least Concern	Least Concern	3597	380	1445	5422
Common Whimbrel	*Numenius phaeopus*	Least Concern	Least Concern	0	0	3	3
Curlew Sandpiper	*Calidris ferruginea*	Near Threatened	Near-threatened	6	0	0	6
Egyptian Goose	*Alopochen aegyptiaca*	Least Concern	Least Concern	0	0	21	21
Goliath Heron	*Ardea goliath*	Least Concern	Least Concern	0	0	2	2
Greater Flamingo	*Phoenicopterus roseus*	Least Concern	Vulnerable	2599	1707	899	5205
Grey Heron	*Ardea cinerea*	Least Concern	Least Concern	1	1	128	130
Greyheaded Gull	*Chroicocephalus cirrocephalus poiocephalus*	Least Concern*	Least Concern	0	6	7	13
Hartlaub's Gull	*Chroicocephalus hartlaubii*	Least Concern*	Vulnerable	362	147	539	1048
Kelp Gull	*Larus dominicanus vetula*	Least Concern*	Least Concern	1148	568	1546	3262
Kittlitz's Plover	*Charadrius pecuarius*	Least Concern	Least Concern	85	22	82	189
Lesser Flamingo	*Phoeniconaias minor*	Near Threatened	Vulnerable	358	314	900	1572
Little Egret	*Egretta garzetta*	Least Concern	Least Concern	0	0	52	52
Little Grebe	*Tachybaptus ruficollis*	Least Concern	Least Concern	0	0	2	2
Marsh Sandpiper	*Tringa stagnatilis*	Least Concern	Least Concern	0	0	1	1
Pied Avocet	*Recurvirostra avosetta*	Least Concern	Least Concern	1668	64	137	1869
Pied Kingfisher	*Ceryle rudis*	Least Concern	Least Concern	2	0	18	20
Ringed Plover	*Charadrius hiaticula*	Least Concern	Least Concern	1	0	2	3
Sacred Ibis	*Threskiornis aethiopicus*	Least Concern	Least Concern	0	6	20	26
Sanderling	*Calidris alba*	Least Concern	Least Concern	738	65	224	1027
Sandwich Tern	*Thalasseus sandvicensis*	Least Concern	Least Concern	85	8	69	162
South African Shelduck	*Tadorna cana*	Least Concern*	Least Concern	144	4	50	198
Swift Tern	*Thalasseus bergii bergii*	Least Concern*	Least Concern	2202	341	258	2801
Threebanded Plover	*Charadrius tricollaris*	Least Concern	Least Concern	0	0	1	1
Whitebreasted Cormorant	*Phalacrocorax lucidus*	Least Concern*	Least Concern	1201	30	416	1647
Whitefronted Plover	*Charadrius marginatus*	Least Concern	Least Concern	37	5	40	82
Grand Total				35181	4166	13061	52408

[1] May include some Arctic Terns, *S. paradisaea*

The Pearson correlation coefficient between salinity (data derived from Chapter 1) and bird counts per pond was negative, but nonsignificant (r = -0.054, df = 27, P>0.2). The regression of bird counts against age of ponds (based on the data in Appendix 3.1) was positive (y = 16.905x + 349) but marginally non-significant (R^2 = 0.0336, df = 92, p = 0.054). Conservatively this can be interpreted as showing that there was no decline in bird numbers as ponds aged and became more saline. There was no significant relationship between numbers of bird species per pond and the age of ponds (R^2 = 0.0033, df = 92, p > 0.2).

Community Composition:

There were significant differences in species composition among areas, based on species present and relative abundance (PERMANOVA, $F_{2,92}$ = 5.535, p = 0.0001), and post-hoc pair-wise comparisons showed that middle and south ponds did not differ, but other comparisons were significant (Table 3.2).

Table 3.2: One-way PERMANOVA of species composition among north, middle and south areas. ***Indicates significant differences at p < 0.001

Area	South	Middle	North
South	0	0.1631	0.0001***
Middle	0.1631	0	0.0001***
North	0.0001***	0.0001***	0

Of the 36 species, 23 were present in the North and Middle areas and 35 in the South area (Table 3.3). Measures of diversity (species richness and Shannon Index) were greatest in the south, although evenness peaked in middle ponds.

Table 3.3: Diversity indices for North, Middle and South

	North	Middle	South
No. of species	23	23	35
Shannon	1.837	2.022	2.114
Evenness	0.273	0.3285	0.2366

Similarity Percentage (SIMPER) identified the main species distinguishing among North, Middle and South using the Bray-Curtis similarity measure. Appendix 3.2 (p117) lists the species in order of importance.

Out of the 36 species, eight were cumulatively responsible for 81.25% of the contribution in North, Middle and South (Fig. 3.2). The clearest patterns were that (1) Blacknecked Grebes were dominant in the north and rarely recorded elsewhere; (2) Cape Cormorants were a major constituent in the south and only occasionally seen elsewhere; (3) Greater flamingos occurred in all three areas but were prevalent in the middle ponds.

Ten species, mostly waders, were found only in the South area: African Black Oystercatcher, African Spoonbill, Blacksmith Plover, Common Whimbrel, Egyptian Goose, Goliath Heron, Little Egret, Little Grebe, Marsh Sandpiper and Threebanded Plover (Table 3.1). During the study, a number of southern ponds were being 'dewatered' and as the water levels slowly dropped, benthic invertebrates were exposed, attracting various foraging waterbird species; and this was probably the reason why more species were present in the Southern ponds.

The middle ponds had recently been mined or were being actively mined and had the lowest bird counts, with the Greater Flamingo being most abundant and the only species with a count of over 1000 (Table 3.1). The northern ponds, where all mining activities ended more than 15 years previously, so that there is now no disturbance to birds, had the greatest counts, with the Blacknecked Grebe, Common Tern, Greater Flamingo, Swift Tern, Pied Avocet, Cape Teal, Kelp Gull, Whitebreasted Cormorant and Cape Cormorant being most abundant with counts over 1000 each (Table 3.1).

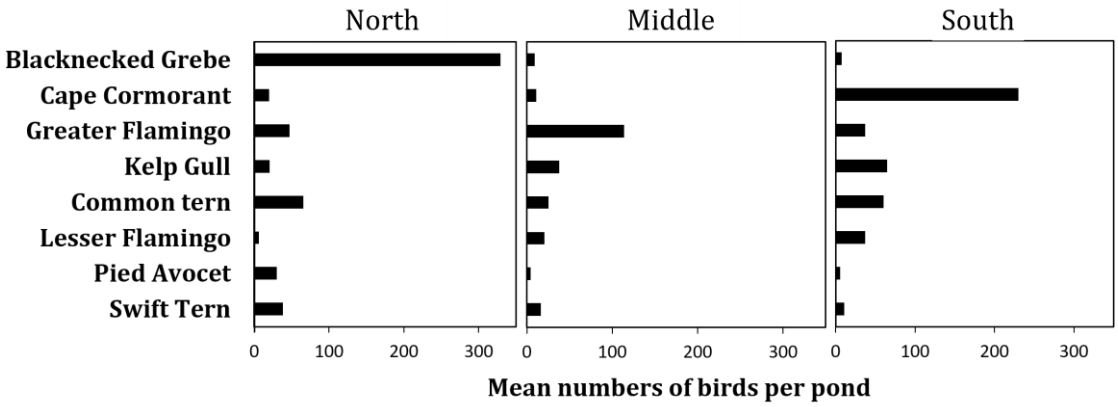


Figure 3.2: Mean abundances of the eight species contributing to the top 81.25% of the cumulative difference among of areas

Pond similarity

The hierarchical clustering dendrogram of community composition was based on the similarity amongst waterbird species per pond using Ward's similarity linkage: ponds with the most similar species are clustered together, and similar clusters are again joined into superclusters (Fig. 3.3).

Six major clusters emerged from the Cluster analysis:

a) Cluster 1 contained a group of nine North ponds in which overall numbers of birds were very high (spanning 1312 to 3522 per eight counts), numbers of species were high (6-16 per pond per count); and Blacknecked Grebes were clearly dominant (accounting for 68% of the 14761 birds counted at ponds in this cluster).
b) Cluster 2 comprised a single pond (N84) in which Common Terns and Swift Terns were most abundant, constituting 81% of the total count of 4186 birds – higher than at any other pond.
c) Cluster 3 covered six South ponds, the majority of which were recently dewatered, where Cape Cormorants were most abundant (51% of the total count of 9734 for the cluster); but other common species included Greater Flamingo and Kelp Gull. The African Black Oystercatcher also occurred more commonly in this cluster than elsewhere (81 of the total count of 87 for this species).
d) Cluster 4 contained 12 North ponds with a relatively high diversity of species (5-14 species per pond) but moderate overall numbers (365-874 birds per pond over the eight counts). As with other north ponds, Blacknecked Grebes were again dominant (68% of the 7027 birds counted for the cluster), but there were also notable abundances of Cape Teal (3.9%), Greater Flamingo (8.7%), Pied Avocet (2.6%) and Whitebreasted Cormorant (4.3%).
e) Cluster 5 was a single Middle pond (M33), typified by an abundance of Greater Flamingos (74%) and Cape Teal (9.2%) out of a total of 1291 birds.
f) Cluster 6 was divisible into several sub-clusters, related in part to area and in part to the extent of mining activities taking place. Sub-clusters 6a and 6b were almost exclusively north ponds with moderately high numbers of birds (166-1146 per pond for the eight counts summed) and a high diversity of species (up to 13 per pond). Whereas subcluster 6a was predominated by Common Terns (34% of a total of 3491 birds), in subcluster 6b Blacknecked Grebes were most prevalent (43% of 4462

birds). Subcluster 6c was a small group of 3 ponds from middle and south dominated by both species of flamingo (59% of 2177 birds). Subcluster 6d united a large number of ponds with low numbers (0-157) and species richness (0-7), including six South ponds that were being actively mined and had no birds (S8, S12, S13, S18, S19, S21); 6e and 6f comprised a mix of ponds from all three areas, 6e dominated by Blacknecked Grebe (31%) and Greater Flamingo (27% of 2095 birds); and 6f by Cape Cormorant (23%), Kelp Gull (24%) and Swift Tern (13% of 2573 birds).

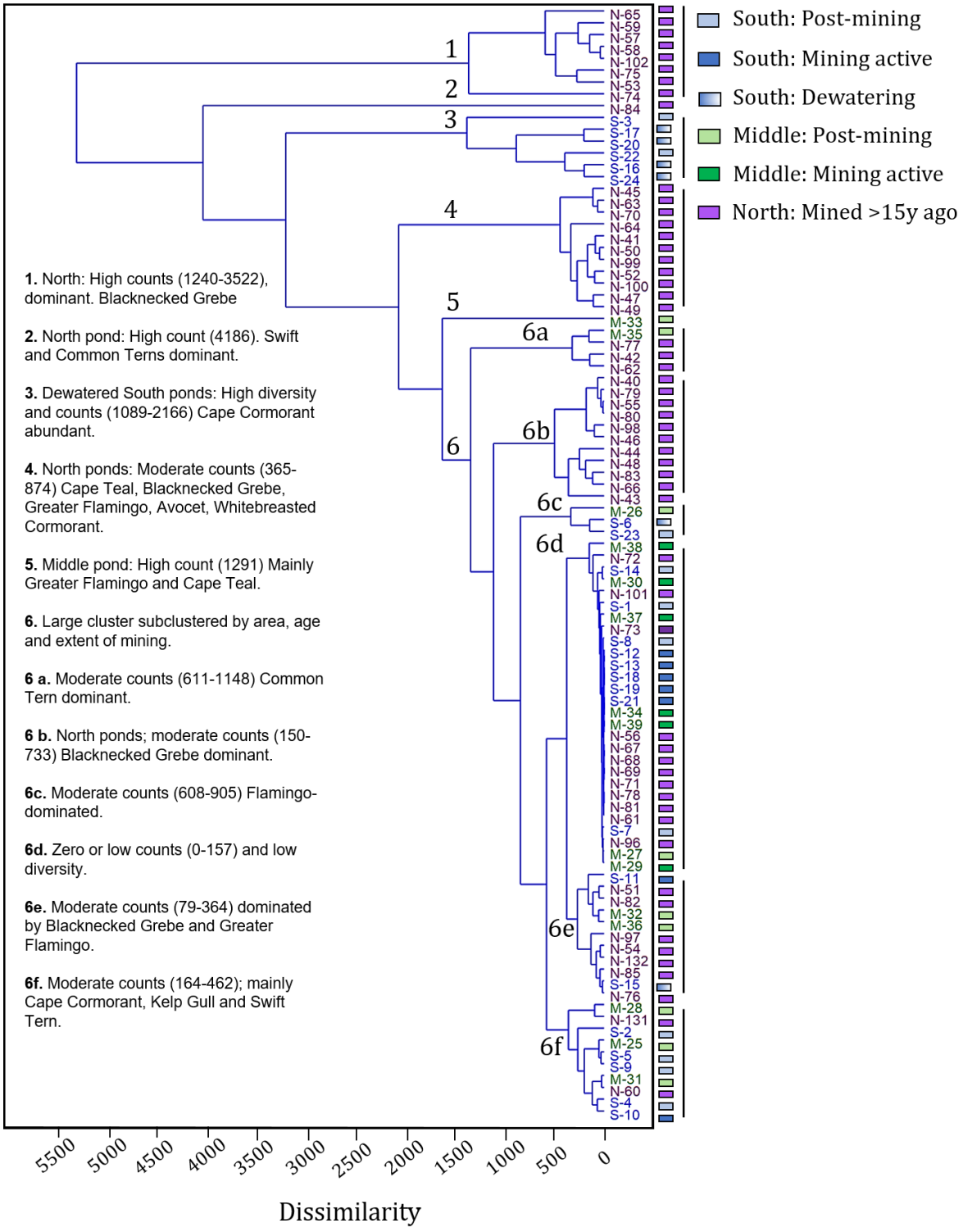


Figure 3.3: Dendrogram representing the similarity amongst waterbird species per pond

Mining activities

The study area was divided into three sections according to the current mining activities, namely south (dewatering and active mining), middle (active and recently mined areas) and north (no recent mining activities). As no active mining was taking place in the North area, the total bird counts from the southern and middle ponds were first plotted and analysed separately according to the activities that were current during the surveys, followed by a consideration of the ponds in the north.

Mining areas and activities were divided into four categories, namely:

1. Dewatering (area being dewatered to allow for mining).
2. During mining (stripping of overburden material and active mining taking place).
3. Post mining, young ponds: areas mined out recently, but more than 1 year ago, which have since flooded or are busy flooding, through natural seepage.
4. Post-mining, old ponds: areas that were mined >15 years ago, all of which lay in the north.

One-way analyses of variance revealed significant differences in bird counts among the phases of mining in the south and middle ($F_{2,36} = 14.90$, $p < 0.0001$). During dewatering, bird counts were generally high, averaging 1266.6 birds per eight counts (Fig. 3.4a), The highest bird count in the south and middle, 2166, was from a southern pond (S20) that was being dewatered. The six dewatered ponds (S6, S15, S16, S17, S20, S24) accounted for 44% of the total of 17227 birds counted for the 39 South and Middle ponds. All except two dewatering ponds (S6 and S15) were in cluster #3, with similar abundances and composition (Fig. 3.3).

The average count from recently mined areas was 488.8 per pond from the 17 post-mining ponds where mining ended more than a 1 year ago and the areas were (or are) busy filling up naturally through seepage (Fig. 3.4a). These included 11 southern ponds (S1, S2, S3, S4, S5, S7, S8, S9, S14, S22, S23) and six middle ponds (M26, M27, M32, M33, M35, M36). Of these post-mining ponds, one stood out as having the highest count: S3, where mariculture activities take place, and where the species composition of birds was similar to that in the dewatered ponds in cluster #3.

The lowest counts, averaging 82.5 were recorded in areas where mining was active (S10, S11, S12, S13, S18, S19, S21, M25, M28, M29, M30, M31, M34, M37, M38, M39). These ponds were also in the last cluster. No birds were present during active mining activities in S12, S13, S18, S19, S21, M34 and M39.

In the north, where mining ceased >15 years ago, bird numbers were not as high as in the dewatering phase of the south and middle ponds, averaging 639.6 birds per pond for the eight counts summed, in comparison with a mean of 1266.6 for dewatered ponds, but the numbers were substantially greater than the counts for ponds with active mining (mean 82.5), and slightly greater than for the post-mining phase of south and middle ponds (mean 488.8). Considerable variability in counts existed in the north, with three ponds having the highest overall counts, but many supporting no birds at all (Fig. 3.4b).

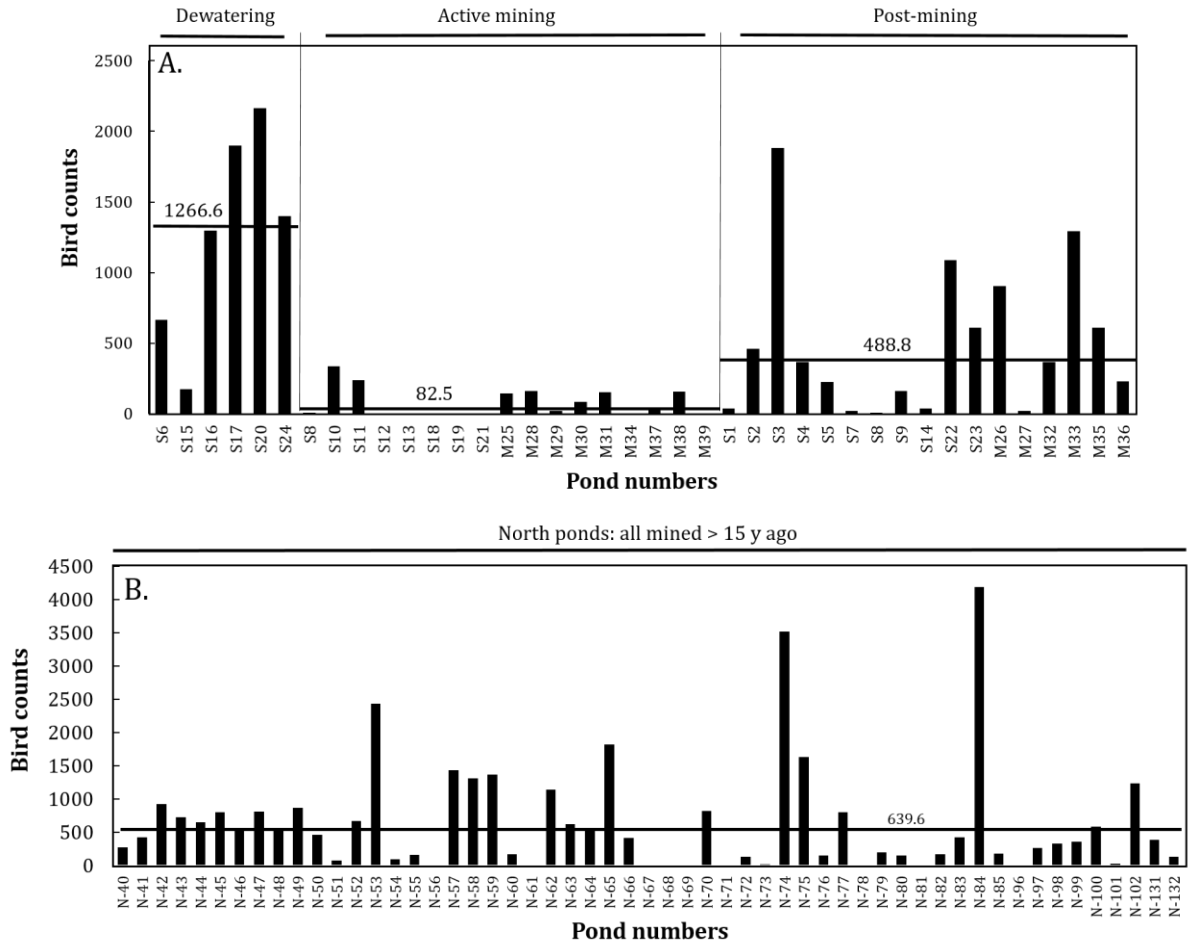


Figure 3.4: Numbers of birds per pond (summed over eight counts) for a): middle (M25-M39) and south (S1-S24) ponds that were being dewatered, actively mined or were in a post-mining state, and b): north ponds. Horizontal bars and values indicate means for different phases

Seasonal affects

Limited seasonal changes in community composition and species abundance were evident. ANOVAs (of summed counts for each of the eight most-abundant species across all three areas) showed Blacknecked Grebe ($F_{3,154} = 1.57$, $p = 0.19$), Cape Cormorant ($F_{3,113} = 0.89$, $p = 0.44$), Greater Flamingo ($F_{3,166} = 1.56$, $p = 0.20$), Common Tern ($F_{3,40} = 0.61$, $p = 0.60$), Pied Avocet ($F_{3,132} = 0.49$, $p = 0.68$), Swift Tern ($F_{3,25} = 0.60$, $p = 0.61$) and Lesser Flamingo ($F_{3,53} = 0.14$, $p = 0.93$) counts were not significantly different among seasons (Fig. 3.5a-g). Only the ANOVA for Kelp Gull revealed a significant seasonal effect ($F_{3,224} = 3.07$, $p = 0.02$) with posthoc tests indicating that numbers were greater in Autumn than Spring, but no other significant differences (Fig. 3.5h).

To explore seasonal patterns graphically for each of the areas separately, the five most abundant species in each area were plotted over seasons (Fig. 3.6a, b, c).

Combined counts in the North were greatest in the summer due to increased numbers of Blacknecked Grebe, Common Tern and Greater Flamingo. The Middle areas also had the highest counts in summer due to increased numbers of Common Tern, Greater and Lesser Flamingos, but declined during winter to spring. The South area had the highest overall counts in spring due to the presence of just over a thousand Common Terns at dewatering ponds during September and November.

The South area was most abundant in Cape Cormorants until their numbers started dropping after winter, coinciding with the dewatering activities, particularly at S17. As this southern pond was dewatered, the central island, mostly inhabited by Cape Cormorants, became connected to the surrounding land, causing the Cape Cormorants to migrate elsewhere (Fig. 3.7).

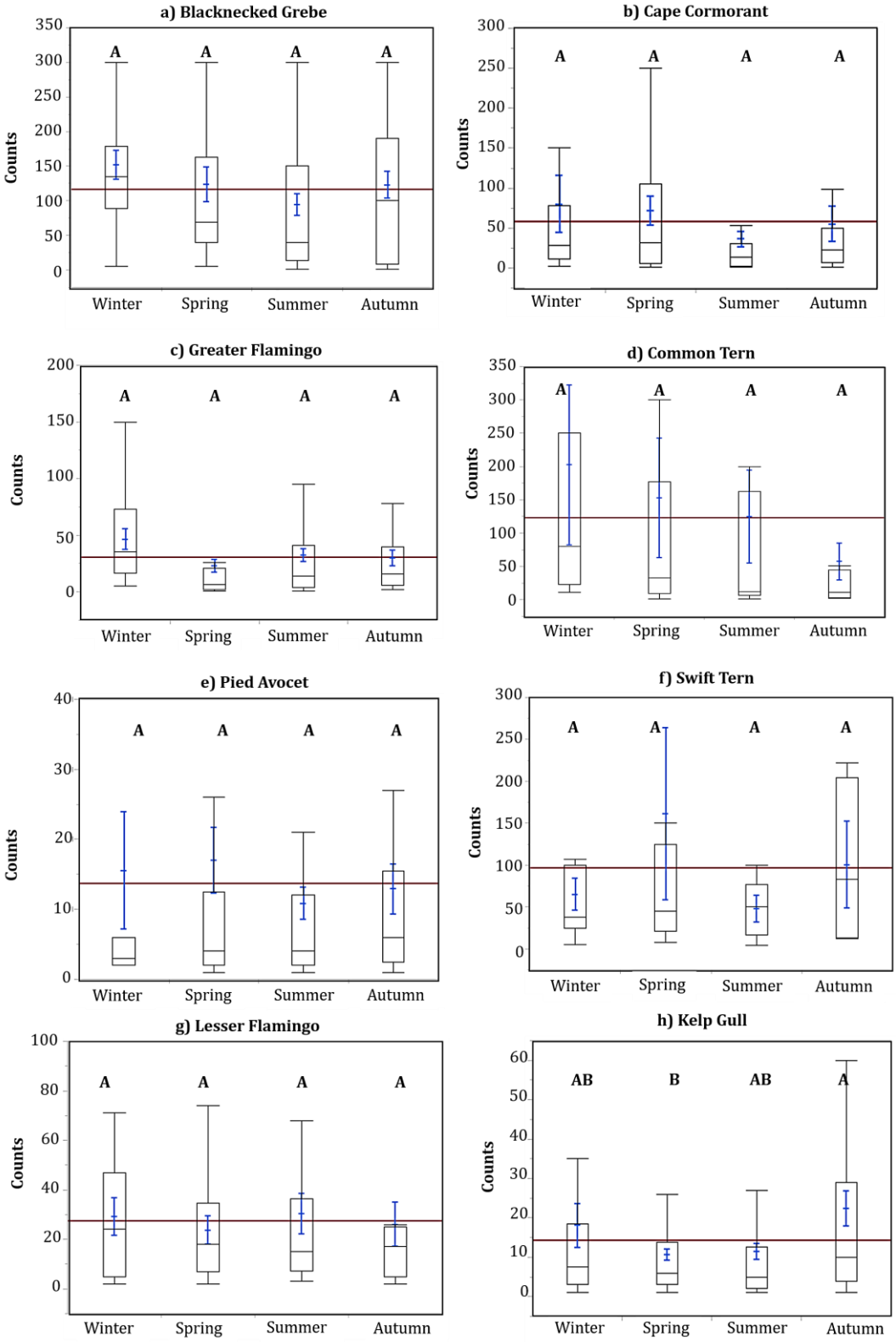
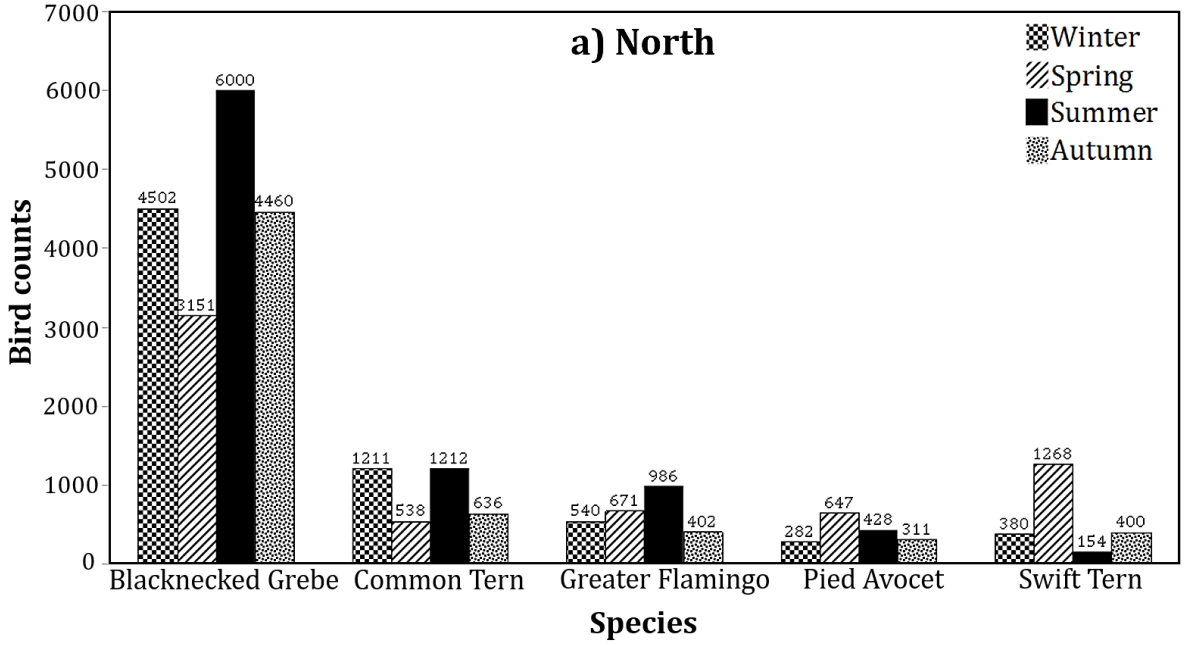
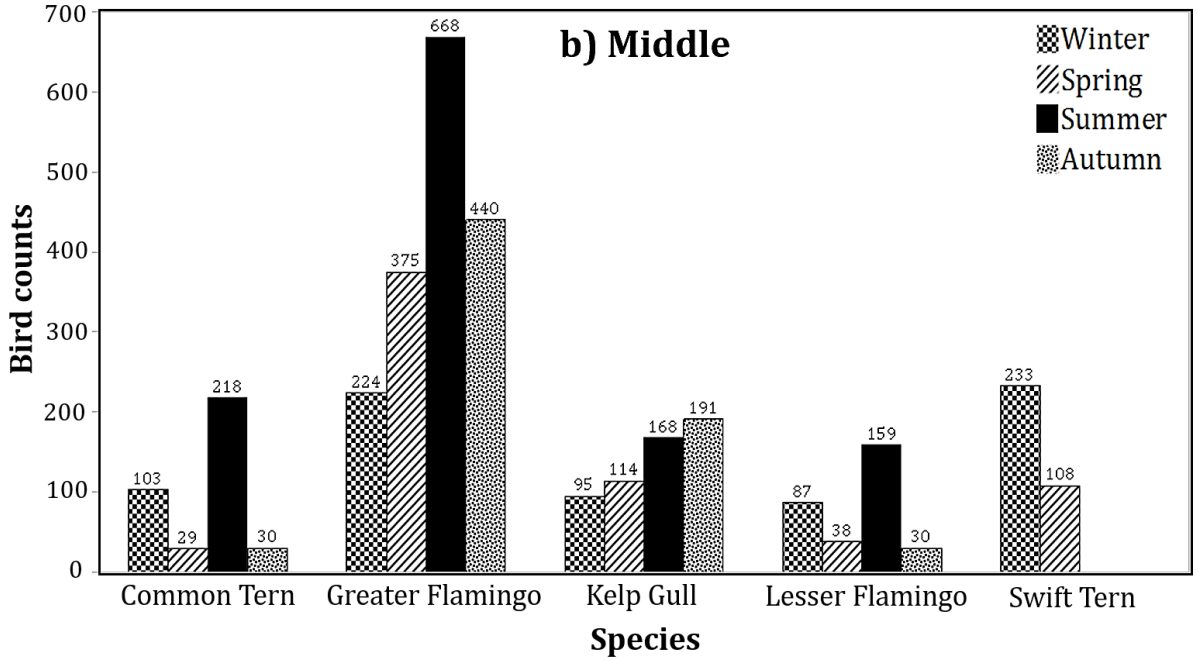


Figure 3.5: Means of top 8 bird species (overall counts from all areas) for a) Blacknecked Grebe, b) Cape Cormorant, c) Greater Flamingo, d) Common Tern e) Pied Avocet, f) Swift Tern, g) Lesser Flamingo, h) Kelp Gull. The red horizontal lines show the grand means and the blue whiskers inside the quartile boxes display the mean error bars. Black box and whisker plots show median values. Similar uppercase letters inside the graphs denote homogeneity between seasons as calculated by post-hoc Tukey tests.


a) North
Winter
Spring
Summer
Autumn
Bird counts
7000
6000
5000
4000
3000
2000
1000
0
4502
3151
6000
4460
1211
538
1212
636
540
671
986
402
282
647
428
311
380
1268
154
400
Blacknecked Grebe
Common Tern
Greater Flamingo
Pied Avocet
Swift Tern
Species


b) Middle
Winter
Spring
Summer
Autumn
Bird counts
700
600
500
400
300
200
100
0
103
29
218
30
224
375
668
440
95
114
168
191
87
38
159
30
233
108
Common Tern
Greater Flamingo
Kelp Gull
Lesser Flamingo
Swift Tern
Species

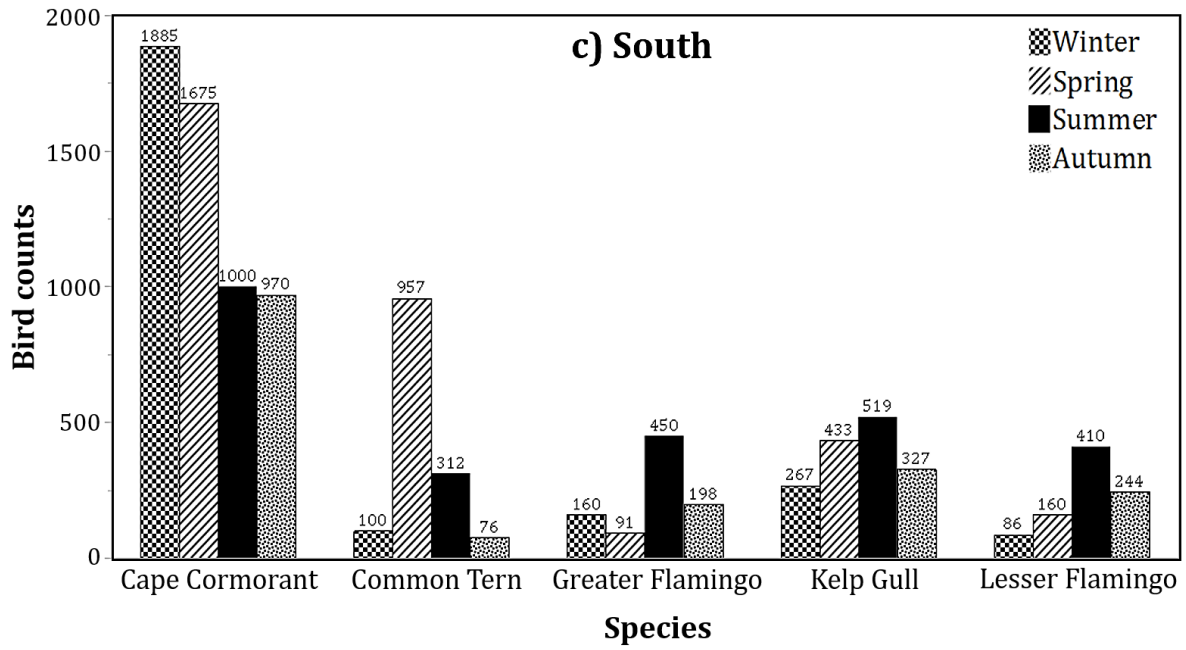


Figure 3.6: Top 5 species per area over seasons (summed over eight counts) for a) North b) Middle c) South

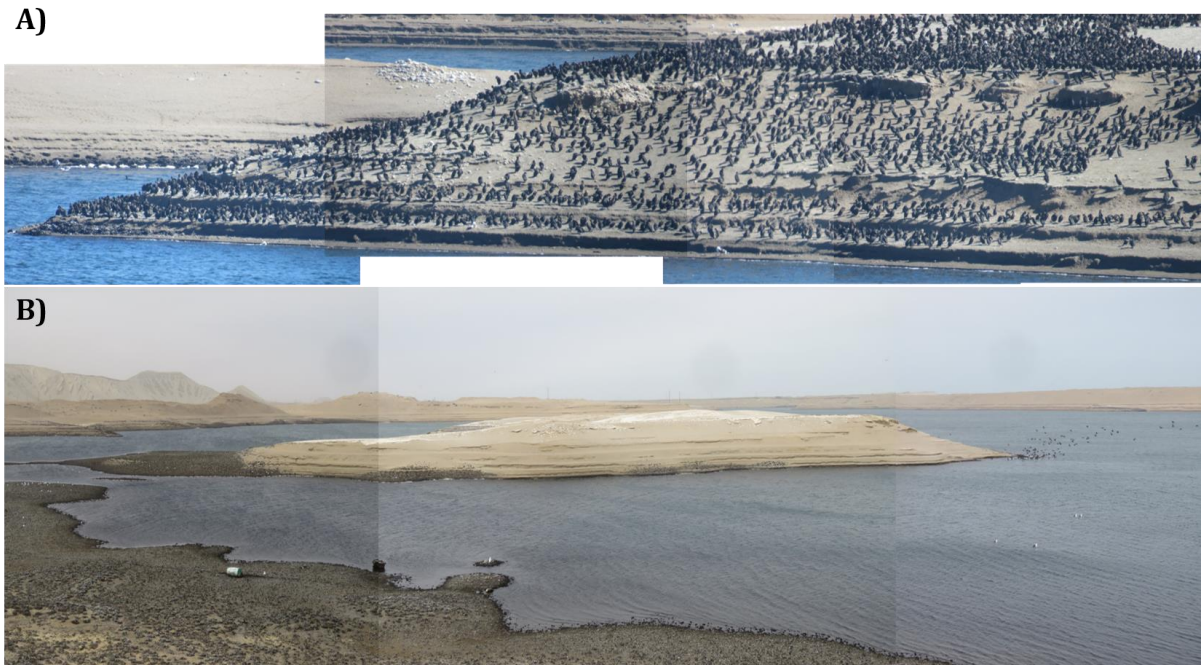


Figure 3.7: Dewatering at S17: a) Island populated mostly with Cape Cormorants; b) Island abandoned due to lowered water levels connecting the island to the land, providing access to scavengers (Jackal and Brown Hyena)

Nests, chicks and juveniles

There were 114 chicks and 709 juveniles in the total count of 52408 birds in the study area (Fig. 3.8). In total, 332 nests belonging to five species were recorded (Fig. 3.8a), with Whitebreasted Cormorants accounting for the lion's share.

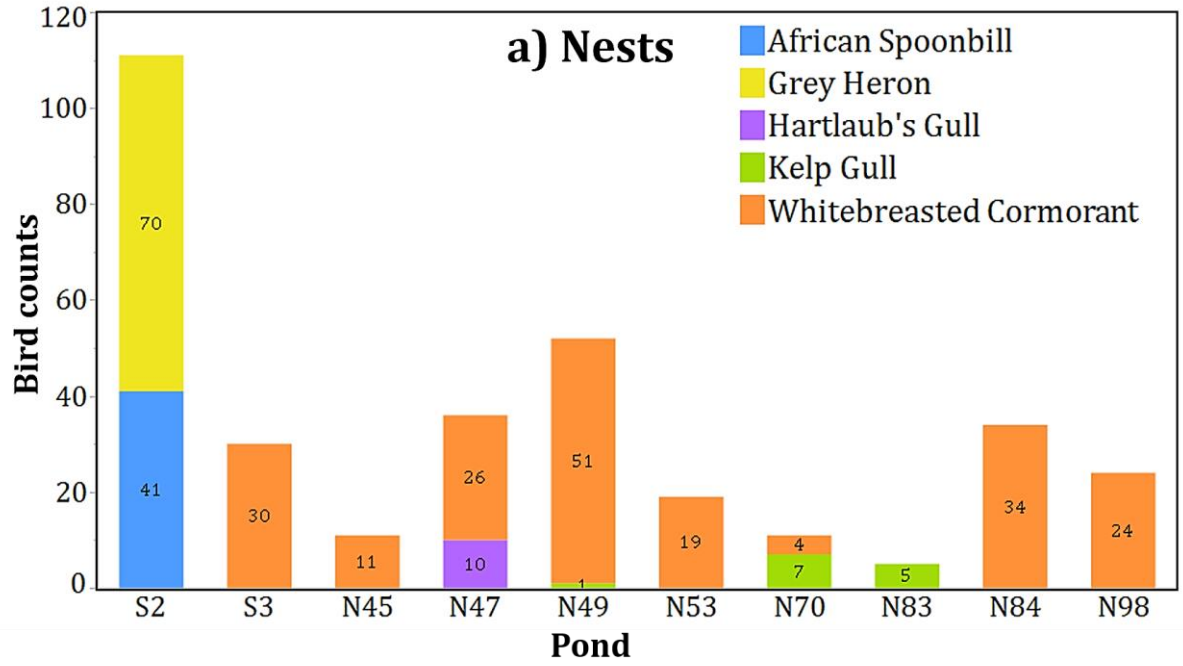
One southern pond (S2) was utilised for nesting and rearing African Spoonbills *Platalea alba* (41 nests, 20 chicks and 9 juveniles) and Grey Herons *Ardea cinerea* (70 nests, 8 chicks and 20 juveniles). The African Spoonbills nested on a small boulder island inside the pond, the Grey Herons nested on an abandoned piece of mining equipment standing next to the pond (Fig. 3.9a, b). Both species reused the same nests from July 2017 to June 2018.

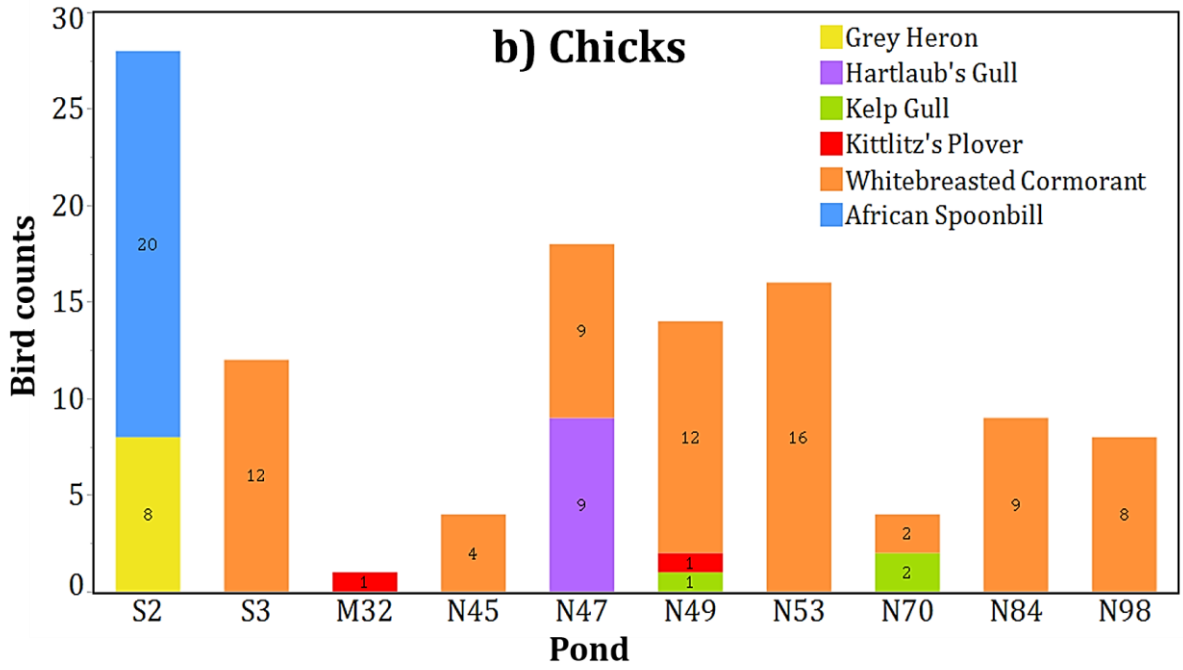
Old powerline poles established next to a southern pond (S3, the pond used for Mariculture activities) were used by Whitebreasted Cormorants *Phalacrocorax lucidus* (30 nests, 12 chicks and 7 juveniles) throughout the study period (Figs. 3.8, 3.9c). The majority of the Whitebreasted Cormorants bred near northern ponds, also on old powerline poles left behind in the area, with a total of 169 nests, 60 chicks and 98 juveniles (Fig. 3.8).

Small rocky islands inside one northern pond (N47) were utilised by Hartlaub's Gull *Chroicocephalus hartlaubii* to build 10 nests and rear 9 chicks (Figs. 3.8, 3.9d). The Kelp Gull *Larus dominicanus vetula* nested in only three northern ponds during the study, N49 with 1 nest, 1 chick; N70 with 7 nests, 2 chicks and N83 with 5 nests (Figs. 3.8, 3.9e), but juveniles were observed in three ponds spanning all areas (S:112, M:24 and N:23).

Neither species of flamingo nested or bred at the ponds, but Greater Flamingo *Phoenicopterus roseus* juveniles were present in all areas (S:145, M:93 and N:86) whereas Lesser Flamingo *Phoeniconaias minor* juveniles were found only in the South (53 juveniles) and Middle (11 juveniles) areas (Figs. 3.8c, 3.9f).

The Middle area was striking for the near-absence of any breeding activity, with only one Kittlitz's Plover chick (M32) being observed in the area.


a) Nests
African Spoonbill
Grey Heron
Hartlaub's Gull
Kelp Gull
Whitebreasted Cormorant
Bird counts
120
100
80
60
40
20
0
70
41
30
11
26
10
51
1
19
4
7
5
34
24
S2
S3
N45
N47
N49
N53
N70
N83
N84
N98
Pond


b) Chicks
Grey Heron
Hartlaub's Gull
Kelp Gull
Kittlitz's Plover
Whitebreasted Cormorant
African Spoonbill
Bird counts
30
25
20
15
10
5
0
20
8
12
1
4
9
9
12
1
1
16
2
2
9
8
S2
S3
M32
N45
N47
N49
N53
N70
N84
N98
Pond

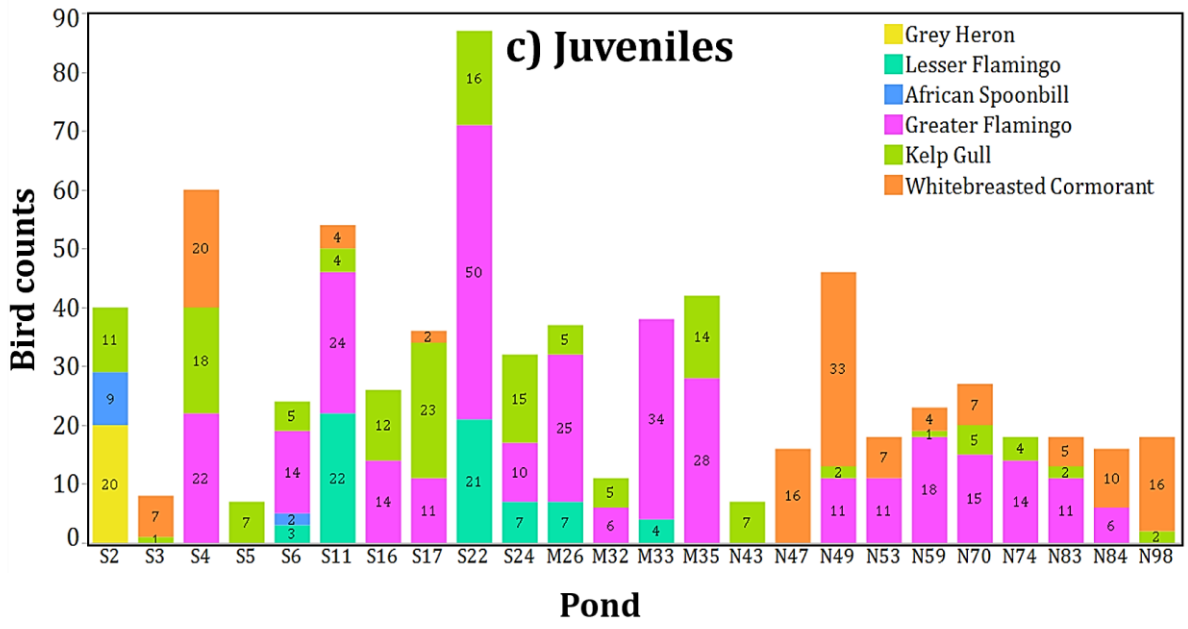


Figure 3.8: a) Nests b) Chicks c) Juveniles present at ponds, summed over the eight counts for the whole year of observation

Figure 3.9: a) Grey Herons nesting on abandoned mining equipment, b) African Spoonbills nesting on a small boulder island, c) Whitebreasted Cormorants nesting on old powerline poles, d) Hartlaub's Gull nesting and rearing chicks on small rock island, e) Kelp Gull rearing chick on small rock island, f) Lesser Flamingo and Greater Flamingo juveniles

DISCUSSION

Unlike the Orange River Estuary, which is a Ramsar Site and has been extensively surveyed in terms of its avifauna (Williams 1986; Simmons and Allan 2002; Anderson et al. 2003), no complete bird counts have been conducted prior to this survey in the restricted diamond area where my study took place. Islands supporting seabirds in the Namibian Islands' Marine Protected Area are found just offshore, but the next closest Ramsar wetland site, Sandwich Harbour (south of Walvis Bay, Namibia), is 500 km to the north and is the nearest comparable natural wetland; and the wetlands of the Olifants River mouth (South Africa) fall 400 km to the south of the study area (Kolberg 2002).

About 55 km north of Sandwich Harbour, lies Walvis Bay, another Ramsar Site and an Important Bird Area, which supports median numbers of birds ranging from 156000 in summer to 82000 in winter (Wearne and Underhill 2005).

The Benguela System, which falls within the Atlantic Ocean, lies parallel to the study site (Sakko 1998; Currie et al. 2008). These nutrient-rich waters, the nearby Orange River Estuary, the Namibian Marine Protected Islands and the mining ponds collectively allow for safe nesting, feeding and roosting sites for a large number and great diversity of seabird and wetland bird populations (Robertson et al. 2012).

The Namibian islands provide important breeding and roosting sites for 10 of 14 seabird species that breed in Namibia, namely: African Penguin (*Spheniscus demersus*), Cape Gannet (*Morus capensis*), Bank Cormorant (*Phalacrocorax neglectus*), Crowned Cormorant (*Microcarbo coronatus*), Cape Cormorant (*Phalacrocorax capensis*), Whitebreasted Cormorant (*Phalacrocorax lucidus*), Kelp Gull (*Larus dominicanus vetula*), Hartlaub's Gull (*Chroicocephalus hartlaubii*), Swift Tern (*Thalasseus bergii bergii*) and African Black Oystercatcher (*Haematopus moquini*) (Currie et al. 2008). The mining ponds supports seven of these seabird species, i.e., Bank Cormorant, Cape Cormorant, Whitebreasted Cormorant, Kelp Gull, Hartlaub's Gull, Swift Tern and African Black Oystercatcher.

The Orange River Estuary supports an average of about 6800 waterbirds of, on average, 52 species (Anderson et al. 2003), including significant proportions of the world's populations of Cape Cormorant and Hartlaub's Gull, both of which are endemic to southern Africa. In addition, it supports more than 1% of the southern African populations of seven other species: Damara Tern, Blacknecked Grebe, Lesser Flamingo, Chestnut Banded Plover, Curlew Sandpiper, Swift Tern and Caspian Tern. The numbers, but not species richness, of waterbirds at the Orange River mouth has decreased since it was designated as a Ramsar Site in 1991. This resulted in the site not meeting one of the four Ramsar criterion under which it was originally designated, namely, that the site no longer regularly supports in excess of 20000 waterbirds, primarily because of the decline in the numbers of Cape Cormorant and Common Tern (Anderson et al. 2003).

Of the 36 species present around the mining ponds, five are endemic to southern Africa (African Black Oystercatcher, Bank Cormorant, Cape Cormorant, South African Shelduck, Hartlaub;s Gull), four are endemic subspecies (Blacknecked Grebe, Swift Tern, Kelp Gull and Greyheaded Gull) and one is an endemic to sub-Saharan Africa (Whitebreasted Cormorant) (Kemper et al. 2007; Sinclair et al. 2011; Simmons et al. 2015). Of these endemics, five are listed in Namibia's Red Data book as being of concern: African Black Oystercatcher, Bank and Cape Cormorants, Blacknecked Grebe and Hartlaub's Gull (Simmons et al. 2015)

Endemics and Red-Listed Species

Cormorants

The Bank Cormorant (*Phalacrocorax neglectus*) and Cape Cormorant (*Phalacrocorax capensis*) are both listed as 'endangered' (Simmons et al. 2015). The Bank Cormorant is the most threated, and its population has more than halved since 1972 (Kemper et al 2007; Crawford et al. 2018); only 125 were encountered during this study. Cape Cormorant numbers have also decreased over the last 30 years (Kemper et al. 2007; Crawford et al. 2007, 2018). Their decline has been particularly severe on the west coast from the Orange River to Dassen Island, accompanied by a southeast movement of the species following a corresponding shift of its forage-fish prey species (Crawford et al. 2016). Of the 6787 I counted over the eight counts, the southern ponds contained the large majority, housing 230 per pond. However, these numbers declined once dewatering activities started in S17, connecting to the 'mainland' an island once utilised by large numbers of the species (Fig. 3.7). The Whitebreasted Cormorant *Phalacrocorax lucidus*, the largest of the cormorants, was the ninth-most-abundant species in this study and is listed as 'least concern' (IUCN 2019) as its numbers are stable (Kolberg 2011). Crawford et al. (2013) attribute this stability to the fact that its main food sources are not harvested.

African Black Oystercatcher

The African Black Oystercatcher (*Haematopus moquini*) is listed as 'near threatened' in Namibia and breeds only in Namibia and South Africa, with core Namibian breeding grounds stretching between Lüderitz Bay and Possession Island, which falls with the Namibian Islands Marine Protected Area (Simmons et al. 2015). Because this species feeds mostly on molluscs such as mussels (Underhill 2014) it was exclusively found in

the South area along dewatered ponds where exposed macrobenthos, notably the mussel *Mytilus galloprovincialis*, was present (Fig. 3.10a). It was, however, never abundant, with its total count being 87.

South African Shelduck

The South African Shelduck (*Tadorna cana*) is a Southern Africa endemic listed as least concern (Simmons et al. 2015) and of the 198 counted during this study, 144 occurred in the north area. This species is often found in mudflats filtering for benthic invertebrates, insect larvae and on vegetation such as submerged macrophytes and hydrophytes (Geldenhuys 1977, 1981).

Blacknecked Grebe

The Blacknecked Grebe (*Podiceps nigricollis gurneyi*), a subspecies endemic to southern Africa that is listed as 'near threatened', is found on many parts of Namibia's coast, including Walvis Bay, Lüderitz and the Namibian offshore islands (Simmons et al. 2015). It is concentrated on the coast in a non-breeding phase during droughts, but rapidly moves to inland wetlands to breed when rains fill pans (Dean and Underhill 1997). Its numbers are increasing, having risen about threefold from 1991 to 2009 (Kolberg 2010b). Of the 18425 Blacknecked Grebes counted during my eight surveys, the large majority occurred in the North area (N: 18110, M: 134, S: 178). They gather in flocks during summer months (Broekhuysen and Frost 1968a, b), and were the main contributor to the swelling of overall bird numbers during summer. Blacknecked Grebes feed on small fish, aquatic insects and larvae, crustaceans and molluscs (Dean and Underhill 1997), and their concentration in north ponds is likely due to the abundance of chironomid fly larvae and pupae in the high-salinity ponds there. The numbers recorded per count in the ponds (2303) is about 10% of the maximum count of 23803 recorded in Walvis Bay in July 2008, although that figure is suspect as it exceeds the estimated total population of birds there (Kolberg 2010b). Its numbers in the ponds constitute about 8.7% of the southern African population, which is substantially above the threshold of 1% set for recognition of Important Bird Areas.

Gulls and Swift Terns

Hartlaub's Gull *Chroicocephalus hartlaubii* is an endemic and is listed as 'Least Concern, although its numbers have doubled over the period 1991-2008 (Kolberg 2013; IUCN 2019). It occurred in all three areas and was found breeding in the North. It was often associated with Swift Terns (*Thalasseus bergii bergii*) – a subspecies endemic to southern Africa, whose numbers are stable (Kolberg 2013). The Common Tern *Sterno hirundo* a migrant to the region, occurred throughout the study area but in highly variable numbers due to its flocking habit. Although both terns did not breed in the study area, Hartlaub's Gull often forms mixed roosts with this species (Williams 1990). The Kelp Gull *Larus dominicanus vetula*, listed as 'least concern' (Whittington et al. 2009), is also an endemic subspecies in southern Africa, where it represents 2% of the global Kelp Gull population. In southern Africa it numbered ca 20000 breeding birds in 1976-1981, rose to ca 42000 in 2000-2005 following cessation of controls on the species on islands, but dropped to around 35000 in 2009-2014 due to predation on its chicks by Great White Pelicans *Pelecanus onocrotalus* in the Western Cape (Kolberg 2013; Whittington et al. 2016). It occurred in all three study areas, N:1148, M:568, S:1546. As opportunistic feeders, Kelp Gulls take advantage of artificial environments, scavenging and preying on other birds' eggs and chicks (Whittington et al. 2016). Grey-headed Gulls, a subspecies endemic to large parts of sub-Saharan Africa and Madagascar and listed as 'least concern', was present but scare at the ponds, despite being widespread along the southern Africa coast and inland (McInnes and Allan 2011).

Flamingos

The Greater Flamingo (*Phoenicopterus roseus*) is listed as 'vulnerable' in Namibia, although its numbers have increased by about 50% over the period 1911-2008 (Kolberg 2012), and is considered the most significant intra-African species according to Williams and Velasquez (1997). It was the fourth-most-abundant species during the study (averaging 650 per count). Greater flamingos are often found feeding in shallow water since they frequently forage on brine shrimps (*Artemia* sp.) and chironomid fly larvae (Velasquez and Hockey 1992). However, their numbers per pond were greatest in the middle ponds, not in the north where high salinities are associated with brine shrimp and chironomid larvae. During the study period, the Greater Flamingo numbers (and those of the Lesser Flamingo) steadily declined from summer to winter, due to adults departing

to breed (mainly in East Africa), leaving behind immature and non-breeding adults (Williams and Velasquez 1997).

The Lesser Flamingo, endemic to southwestern Africa, listed as 'vulnerable' due to decreases in its global populations (Moreno-Opo et al. 2012; Simmons et al. 2015), decreased slightly in southern Africa over the period 1991-2008 (Kolberg 2012). Its mean numbers per count in the ponds (196.5) were relatively low but consistent. The Lesser Flamingo prefers a more saline habitat than the Greater Flamingo, feeding in shallow waters and filtering with its bill upside down for cyanobacteria from the water surface and small diatoms from the bottom layers (Simmons et al. 2015). However, in my survey it was concentrated in the south ponds, not in the north where salinities were high.

Relevance of birds for predators

The presence of bird colonies around the mining ponds benefits other species such as the Black-backed Jackal (*Canis mesomelas*) and Brown Hyena (*Parahyaena brunnea*), the latter of which is listed as 'near threatened' on the IUCN's (2019) list of threatened mammal species. Few bird species nested on the ground close to the ponds edges where they would have been subjected to predation. The majority were found nesting inside ponds on rock islands or high on old powerline poles. During the study, the remains of only one dead Greater Flamingo (Fig. 3.9b) and three Cape Cormorants were found.

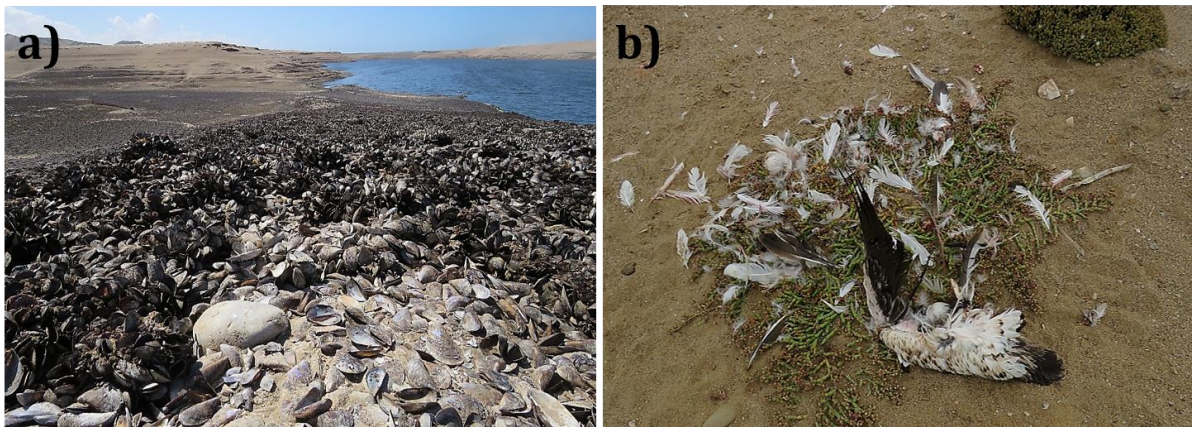


Figure 3.10: a) Exposed mussels (*Mytilus galloprovincialis*) from a dewatered pond, b) Remains of a Greater Flamingo (*Phoenicopterus roseus*) consumed by a predator

Effects of mining

Mining has multiple effects. Firstly, while mining is active, disturbance of birds is at a peak, and all actively mined ponds I monitored had either no birds at all, or very few. Counts and diversity were therefore low in actively mined areas. Secondly, mining may deposit sediments in the sea or on the coast, reducing water visibility. Simmons (2005) has, for example shown that at Elizabeth Bay near Lüderitz, deposition of 'fines' clouds the water, and that the abundance and availability of fish for piscivorous birds have likely been reduced as a result. Coincident with mining activities, the density of shorebirds at Elizabeth Bay has declined 4.3-fold over a 25-y period.

However, a third (and converse) effect of mining is that it can generate ponds that attract birds. Invertebrates become established in the ponds and provide food. The status of these ponds is, however, not static. During mining operations, dewatering lowers the water level, exposing the benthos to foraging birds, and concentrating fish. Cape Cormorants in particular took to feeding in dewatered ponds and became the dominant members of the avifauna in dewatered ponds. Ponds that were being dewatered thus supported large numbers of birds, as Velasquez (1992) has documented for saltpans where water levels are manipulated to harvest salt. After mining ceases, the ponds fill again by seepage and water levels reach a balance between seepage and evaporation. Over time, salinities rise and after around 10-15 years become hypersaline (Chapter 1). However, neither salinity nor the age of ponds were significantly related to bird numbers; it is possible that birds use other water bodies such as the Orange River as a source of drinking water to nullify the high salinities in older ponds.

As water levels change, islands can be created, forming safe habitat for nesting or roosting, or they can be reunited with the surrounding land, making them less suitable for these activities. The numbers of cormorants, in particular, responded to these changes. Numbers rose when islands were formed and declined once the islands ceased to be isolated havens. Similarly, African Spoonbills nested only on islands isolated in the middle of ponds and on the availability of fish for piscivorous birds (Whitfield and Cyrus 1978).

Mining ponds as an alternative habitat

Similar to artificial wetlands and coastal salt ponds, mining ponds can constitute an attractive alternative habitat for waterbirds to nest, forage and roost (Velasquez 1993; Ma et al. 2009; Chokri and Selmi 2011). Globally, many species of waterbirds forage in artificial salt ponds, with shorebirds, ducks, waders, gulls, and terns generally making up the majority of the species (Velasquez 1993). The quality of such artificial ponds as a foraging habitat is largely dependent on the availability of benthic invertebrate fauna to birds, which in turn is determined by water levels and salinity (Velasquez 1993).

Velasquez et al. (1991) examined the relationship between water depth and foraging by waterbirds, by gradually reducing the water levels in artificial saltpans, and found that large waders and shorebirds were attracted to saltpans with lower salinity, whereas smaller waders and shorebirds were attracted to those with higher salinities. This was further confirmed when Velasquez et al. (1992) studied how waterbirds forage in artificial saltpans, saltmarshes and intertidal mudflats during high and low tides at the Berg River estuary in South Africa, when he found that highest foraging densities of waterbirds occurred in salt ponds of 25-70ppt and 170-220ppt (Velasquez 1993).

My measurements of water salinity showed that after remaining stable at salinity levels close to that of seawater for about 10 years, salinity levels then rose with further aging of the mining ponds, with the northern ponds water salinity levels increasing to more than double those in the middle and southern ponds (Chapter 1). Surprisingly, the salinity in the ponds remained relatively close to that of seawater for prolonged periods, likely because of seepage from the sea into ponds and frequent fog events being a constant source of moisture in this arid region (Seely et al. 1998; Henschel and Seely 2008). This explains why higher abundances of certain species were found in certain areas. For instance, Pied Avocet (*Recurvirostra avosetta*) counts were highest in the North area, as they tend to inhabit shallow waters with high salinity levels (Chokri and Selmi 2011).

Salinity also influences vegetation communities. Saltmarshes, comprising a single species, *Salicornia natalensis affinis*, grew around the mining ponds edges, and were used by birds for shelter and nest-building material (Chapter 2). The saltmarshes also can be a food source for birds that feed on vegetation, such as South African Shelduck (*Tadorna*

cana), Egyptian Goose (*Alopochen aegyptiaca*) and Cape Teal (*Anas capensis*) (Russel et al. 2014). Birds play an important role in dispersing seeds along the coast in their guts, feathers and in mud on their feet (Vivian-Smith and Stiles 1994; Neff and Baldwin 2005). Their frequent flights between aquatic communities (van Leeuwen et al. 2012) and their ability to overcome barriers to disperse seeds, preserve connectivity among ponds that might otherwise not be connected (Nathan et al. 2008). Studies have indicated that species richness and abundance of birds increase with vegetation cover in wetlands, which provides food for herbivorous waterbird species and influences habitat use by waterbirds (Ma et al. 2009; Russell et al. 2014).

Colwell and Taft (2000) have shown that water depth plays an important role in influencing the access of waterbirds to benthic food sources, and that this is dependent on each species' morphology, such as leg, bill and neck lengths, which can extend the foraging habitat of non-diving species, allowing them to feed in deeper waters and on smaller benthic invertebrates (Ma et al. 2009). Russell et al. (2014) investigated how water depth influences the accessibility of feeding areas for waterbirds and found mostly waders such as Common Greenshank, Plovers, Common Whimbrel, Sandpipers, African Black Oystercatcher, Little Egret and larger waders such as African Spoonbill and Greater Flamingo feeding in shallower waters such as those experienced in the dewatered ponds, whereas plunge divers (Cape Cormorant and Blacknecked Grebe) can feed in deeper waters.

The distribution and abundance of benthic invertebrate fauna available for waterbirds is also influenced by salinity (Whitfield and Cyrus 1978; Velasquez 1992; Ma et al. 2009). Brine shrimp (*Artemia* sp.) and fly larvae and pupae (Chironomidae) were found in many northern ponds. The Blacknecked Grebe was most abundant in the north area, likely due to higher salinity levels creating favourable environmental conditions for amphipods and chironomid fly larvae on which they feed, whereas the southern ponds, where salinity levels were lower, resembled intertidal mudflats, and are more likely to attract waders (Velasquez 1992).

Inside the middle ponds, insect larvae and pupae (Chironomidae), marine isopods (*Exosphaeroma* spp.), amphipods (*Jassa falcata*) and alien invasive mussels (*Mytilus*

galloprovincialis) were recorded (pers. obs.). The southern ponds, particularly the dewatered ponds, had crown crabs (*Hymenosoma orbiculare*), shrimp (*Palaemon peringueyi*), pipefish (*Syngnathus temminckii*), small fish (gobiid and clinid species), serpent eel (*Ophisurus serpens*), Namibian cushion star (*Asterina stellifera*), Benguela compass jellyfish (*Chrysaora fulgida*) and alien invasive mussels (*Mytilus galloprovincialis*) (pers. obs.; Fig. 3.11). Various fish were found in particular ponds, as described in the next chapter.

While the studies cited above focused on managing water depth, salinity and vegetation for waterbirds in artificial wetlands, Chambon et al. (2019) conducted a study demonstrating the importance of creating habitats (such as artificial islets) that will increase nesting opportunities for waterbirds. His study, in which artificial habitats (islets) were created, successfully increased the numbers of nests for Pied Avocets.

Although the mining ponds cannot replace or substitute a natural wetland, they can provide an alternative habitat for waterbirds (Velasquez 1992). Studies have shown that artificial wetlands, if managed effectively (water levels, vegetation, salinity, benthos availability, wetland connectivity) can reduce the effects of wetland degradation by providing an alternative foraging habitat for waterbirds (Ma et al. 2009).

Figure 3.11: a) South African mullet (*Chelon richardsonii*), b) Brine shrimp (*Artemia* sp.), c) Namibian cushion star (*Asterina stellifera*), d) Benguela compass jellyfish (*Chrysaora fulgida*), e) Shrimp (*Palaemon peringueyi*), f) serpent eel (*Ophisurus serpens*), g) alien invasive mussels (*Mytilus galloprovincialis*), h) crown crabs (*Hymenosoma orbiculare*) that were recorded in middle and south ponds

Evaluating the significance of the ponds to avifauna

The avifauna of the mining ponds can be compared with that of other wetland areas, based on bird counts in the literature. In summary, the ponds supported 36 species including ten endemics, six regionally endangered species and nine species that exceed thresholds specified for Important Bird Areas (IBAs), regional IBAs, 1% levels of the southern African population, or 1% of the flyway population (Table 3.6).

The total number of birds recorded in the ponds was 11026 per count (Fig. 3.12a), which is below the 20000 threshold set for Ramsar sites, and substantially below the average numbers counted for Mile 4 Saltworks (ca. 95000), Walvis Bay (75790) and Sandwich Harbour (69062); but above the counts for the Orange River mouth (9743), Cunene

Mouth (3001) or Baia dos Tigres in Angola (2050), all of which either are, or have been proposed as, Ramsar Sites – and are considered Important Bird Areas (Simmons et al. 1999; Kolberg 2015).

The number of bird species recorded in the ponds (36) exceeded that of four other wetland sites rated as Important Bird Areas, fell within the range of values for four others, and was exceeded only by the range for Cunene Mouth (Fig. 3.12b)

These comparisons are based on total counts, but the dimensions of the sites are very different. A fairer comparison is to express the numbers per km or per km^2 (Fig. 3.12c, d). The ponds rank low when assessed as density.km^{-1}. However, in terms of density.km^{-2}, they are equivalent to Cape Cross lagoon and exceeded by only four of the other sites.

The number of regionally threatened species housed in the ponds (seven) was greater than that in five of the wetlands with which I made comparisons (Fig. 3.12e). The ponds supported more than 1% of the regional population of nine species, exceeding the numbers recorded in five of the nine other wetlands (Fig. 3.12f). Comparing thresholds specified for recognition of Important Bird Areas IBAs), Regional IBAs, Ramsar sites and 1% of the southern African or flyway population (Table 3.6), ponds ranked highly for at least one of these thresholds for nine species, including Blacknecked Grebe which reached abundances an order of magnitude above all five thresholds considered.

Table 3.6: Mean numbers of birds in ponds, expressed per count, compared with thresholds set for Important Bird Areas (IBAs), Regional IBAs, Ramsar sites, and 1% levels for southern Africa populations. Numbers in bold were exceeded by those recorded in the ponds. Species are not listed if their mean counts were < 1.

Species	Mean number per count in ponds	IBA threshold	Regional IBA threshold	Ramsar threshold	1% of Sn African population	1% of flyway population
African Black Oystercatcher	10.9	48	24	47	67	50
African Spoonbill	18.3	150	75	1000	**10**	
Bank Cormorant	18.6	150	75	1000	97	97
Blacknecked Grebe	2303.1	**250**	**125**	**150**	**265**	**150**
Blackwinged Stilt	46.1	1000	500		150	230
Cape Cormorant	848.4	5500	2750	3000	**2340**	2200
Cape Teal	209.8		1750	1750	1750	1750
Common Tern	677.8	13000	8000	4000	3500	6400
Curlew Sandpiper	0.8	7500	3750		1150	3300
Egyptian Goose	2.6	3500	1750	3500	3000	1500
Greater Flamingo	650.6	1250	**625**	760	**550**	750
Grey Heron	16.3				100	
Greyheaded Gull	2.9	1000	500	3000	40	
Hartlaub's Gull	130.1	250	**125**	300	325	300
Kelp Gull	408.6	**300**	**150**	700	**276**	700
Kittlitz's Plover	23.6		500		550	
Lesser Flamingo	196.5	20000	10000	600	1600	600
Little Egret	5.3	1000	500		40	
Pied Avocet	233.6	250	**125**		**150**	**190**
Sacred Ibis	3.3	2000	1000	3300	50	
Sanderling	128.4	2200	1100	1500	850	1200
Sandwich Tern	43.9	1500	750	1700	200	1700
South African Shelduck	24.8	420	210	500	300	
Swift Tern	326.5	500	**250**	**200**	**150**	**200**
Whitebreasted Cormorant	205.9		2500	**150**	**74**	**120**
Whitefronted Plover	10.3	250	125	180	180	190
Total numbers of wetland birds:	11028					
Number of species:	36					

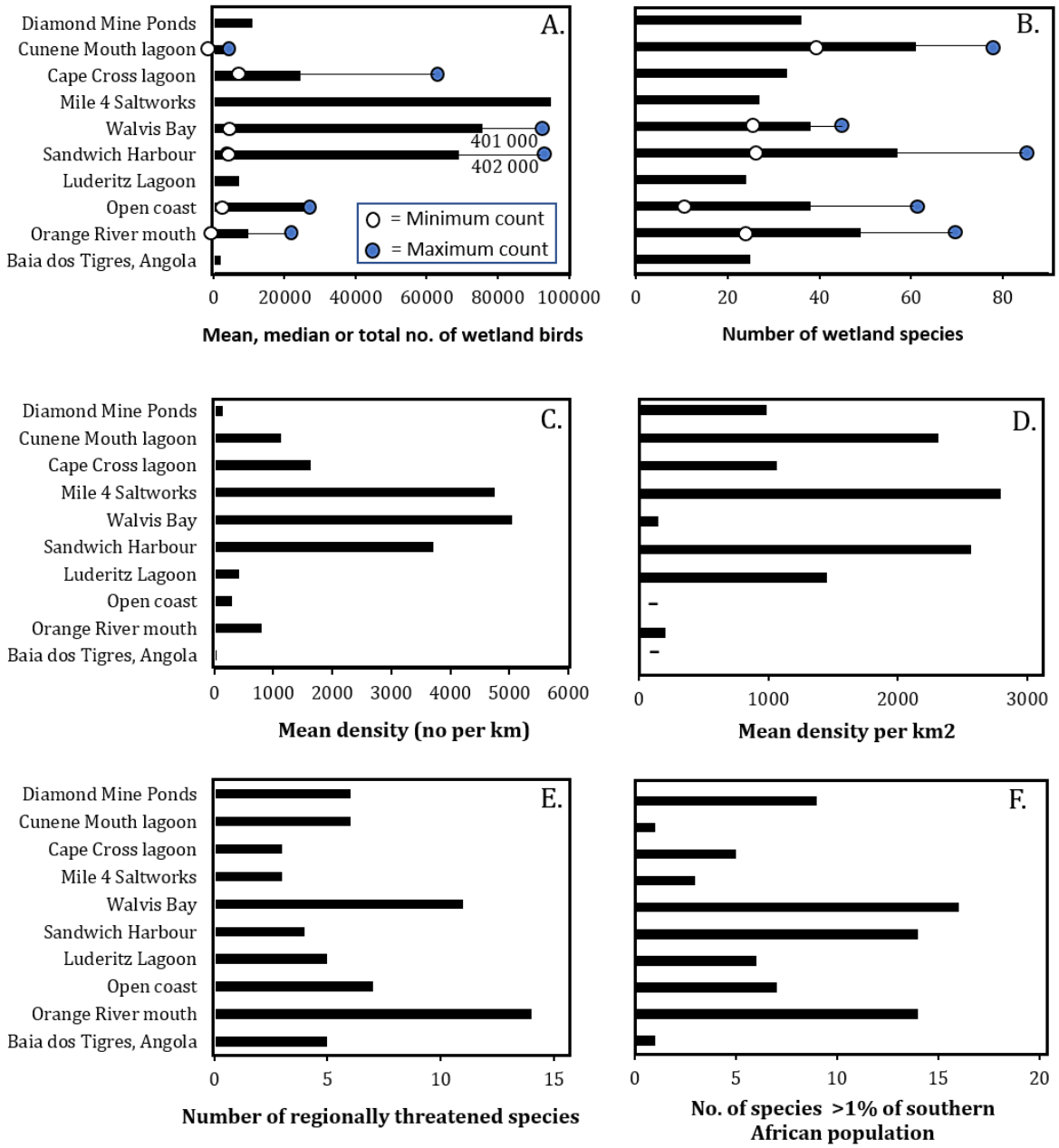


Figure 3.12: Analysis of how the ponds rank relative to other wetlands, evaluated by six criteria (derived from Appendix 3.3.)

CONCLUSION

Diamond mining along the south-western Namibian coastline has created multiple seawater ponds lying immediately adjacent to the coastline. These ponds provide an attractive alternative habitat for waterbirds to roost, forage or breed, particularly because of their consistent water levels (barring times of dewatering), which makes them regular and predictable habitats most of the time (Velasquez 1992; Velasquez and Hockey 1992). Mine is the first survey to quantify the avifauna associated with these ponds. The near threatened Blacknecked Grebe (*Podiceps nigricollis*), endangered Cape Cormorant (*Phalacrocorax capensis*) and Common Tern (*Sterna hirundo*) were among the most abundant species in the study area. The ponds are an important habitat for these species especially since the decline in Cape Cormorant and Common Tern numbers along the west coast in general, and at the Orange River Ramsar site in particular (Anderson et al. 2003; Kemper et al. 2007; Crawford et al. 2007, 2018).

Overall, the ponds can be regarded as an Important Bird Area, even although the total number of birds supported falls short of the Ramsar threshold of 20000 birds. The potential for the ponds to continue to be used by birds will depend on (a) the possibility of creating artificial islands in selected ponds after mining, for nesting and roosting sites, (b) understanding the benthic invertebrate fauna availability, which is important to interpreting future changes in waterbird abundance and occurrence (Colwell and Taft 2000), and (c) the long-term viability of the ponds in the light that they are likely to become shallow and hypersaline after about 15 y. In this latter context, because there was no correlation between salinity and bird numbers, the life of the ponds as a bird habitat may extend beyond the time they begin to experience elevated salinities. Similarly, the absence of any negative effect of pond age on bird numbers or species richness suggests that even very old ponds will remain a viable habitat for birds.

These points all support findings by Velásquez (1992) that wetlands with a wide range of salinities support bird populations, and that habitat changes and water level fluctuations at artificial saltpans are critical in determining the occurrence, diversity and abundance of birds.

In short, the ponds generated by mining do serve as an important habitat for birds, and are likely to continue to do so for considerable periods of time, even without renewed contact with the sea.

The chapter to follow will focus on the extent to which the ponds support another important faunal group, namely the fish populations.

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APPENDICES

Appendix 3.1:
Bird monitoring ponds, listing coordinates, relative size, date last mined (with age in years since mining italicised), mining status (post-mining, during mining or being dewatered), total number of birds counted in eight counts, and total number of species recorded (italicised)

Pond No.	GPS Coordinates	Size	Last Mined	Mining status	Total count *No. of species*
South 1	28°36'1.70"S	Intermediate	3/1/2004	Post	35
	16°24'53.20"E		*14*		*2*
South 2	28°35'39.21"S	Small	12/1/2000	Post	462
	16°24'51.28"E		*18*		*12*
South 3	28°35'42.82"S	Big	12/1/2000	Post	1884
	16°23'43.95"E		*18*		*16*
South 4	28°35'31.34"S	Intermediate	12/1/2014	Post	367
	16°23'28.75"E		*4*		*13*
South 5	28°35'9.50"S	Intermediate	4/1/2015	Post	225
	16°23'1.24"E		*3*		*9*
South 6	28°35'5.96"S	Intermediate	1/1/2017	Dewatering	664
	16°22'36.78"E		*1*		*18*
South 7	28°34'47.84"S	Intermediate	7/1/2016	Post	18
	16°22'31.39"E		*2*		*3*
South 8	28°34'47.41"S	Intermediate	1/1/2013	Post	6
	16°22'31.07"E		*5*		*1*
South 9	28°35'0.20"S	Small	1/1/2013	Post	160
	16°22'25.40"E		*5*		*6*
South 10	28°34'36.24"S	Small	5/1/2017	During	335
	16°21'45.25"E		*1*		*8*
South 11	28°34'47.02"S	Intermediate	9/1/2013	During	239
	16°22'1.58"E		*5*		*9*
South 12	28°34'30.47"S	Small	3/10/2018	During	0
	16°22'14.93"E		*0*		*0*
South 13	28°34'36.02"S	Intermediate	23/10/2018	During	0
	16°21'44.26"E		*0*		*0*
South 14	28°34'20.63"S	Intermediate	1/1/2013	Post	36
	16°21'35.37"E		*5*		*6*
South 15	28°33'17.67"S	Big	5/7/2018	Dewatering	172
	16°20'11.76"E		*0*		*4*
South 16	28°33'3.49"S	Intermediate	9/7/2018	Dewatering	1299
	16°19'54.86"E		*0*		*23*
South 17	28°32'20.71"S	Small	1/7/2018	Dewatering	1898
	16°19'14.75"E		*0*		*16*
South 18	28°32'7.50"S	Intermediate	6/1/2011	During	0
	16°18'52.13"E		*7*		*0*
South 19	28°31'54.67"S	Intermediate	5/1/2011	During	0
	16°19'10.79"E		*7*		*0*
South 20	28°31'43.16"S	Intermediate	7/1/2016	Dewatering	2166
	16°18'28.90"E		*2*		*17*
South 21	28°31'33.84"S	Intermediate	7/1/2013	During	0
	16°18'17.52"E		*5*		*0*
South 22	28°31'20.40"S	Small	7/1/2016	Post	1089

	16°18'16.42"E		*2*		*13*
South 23	28°31'8.45"S	Big	12/1/1993	Post	608
	16°18'7.49"E		*26*		*14*
South 24	28°30'44.61"S	Big	12/1/1993	Dewatering	1398
	16°17'38.15"E		*26*		*17*
Middle 25	28°30'39.13"S	Small	6/1/2018	During	144
	16°17'23.88"E		*0*		*2*
Middle 26	28°30'29.76"S	Small	12/1/1980	Post	905
	16°17'22.26"E		*38*		*14*
Middle 27	28°30'11.68"S	Small	12/1/1980	Post	21
	16°17'5.04"E		*38*		*6*
Middle 28	28°30'25.06"S	Small	8/1/2017	During	163
	16°16'54.08"E		*1*		*4*
Middle 29	28°30'17.78"S	Intermediate	11/1/2016	During	19
	16°16'44.98"E		*1.5*		*4*
Middle 30	28°30'0.21"S	Intermediate	12/1/2016	During	83
	16°16'22.58"E		*1.5*		*7*
Middle 31	28°29'39.91"S	Big	1/1/2016	During	151
	16°16'1.48"E		*1.5*		*7*
Middle 32	28°29'23.41"S	Big	11/1/2015	Post	364
	16°15'44.53"E		*3*		*12*
Middle 33	28°29'3.62"S	Small	12/1/1997	Post	1291
	16°15'30.21"E		*21*		*9*
Middle 34	28°28'36.97"S	Small	1/1/2016	During	0
	16°15'26.40"E		*1.5*		*0*
Middle 35	28°28'24.92"S	Small	12/1/1997	Post	611
	16°15'17.99"E		*21*		*10*
Middle 36	28°28'3.93"S	Big	2/1/2016	Post	228
	16°14'51.58"E		*1.5*		*7*
Middle 37	28°27'32.58"S	Small	8/1/2016	During	29
	16°14'12.06"E		*1.5*		*3*
Middle 38	28°27'35.16"S	Intermediate	12/1/2017	During	157
	16°14'25.32"E		*1*		*7*
Middle 39	28°27'13.34"S	Intermediate	12/1/2017	During	0
	16°13'56.06"E		*1*		*0*
North 40	28°26'42.52"S	Small	8/1/2006	None	275
	16°13'39.05"E		*12*		*5*
North 41	28°26'36.89"S	Small	8/1/2006	None	427
	16°13'32.77"E		*12*		*7*
North 42	28°26'31.75"S	Intermediate	8/1/2006	None	929
	16°13'23.64"E		*12*		*11*
North 43	28°26'22.06"S	Intermediate	8/1/2006	None	733
	16°13'12.26"E		*12*		*9*
North 44	28°26'2.39"S	Intermediate	2/1/2006	None	655
	16°12'55.17"E		*12*		*12*
North 45	28°25'44.79"S	Intermediate	4/1/2004	None	804
	16°12'38.74"E		*14*		*11*
North 46	28°25'33.44"S	Small	2/1/2006	None	545
	16°12'29.69"E		*12*		*10*
North 47	28°25'24.47"S	Intermediate	4/1/2004	None	814
	16°12'18.10"E		*14*		*13*

North 48	28°25'7.36"S	Big	6/1/2004	None	560
	16°11'56.98"E		*14*		*12*
North 49	28°24'43.01"S	Big	6/1/2004	None	874
	16°11'33.66"E		*16*		*14*
North 50	28°24'32.72"S	Small	12/1/2002	None	469
	16°11'27.51"E		*16*		*5*
North 51	28°24'30.12"S	Small	12/1/2002	None	79
	16°11'25.48"E		*16*		*7*
North 52	28°24'25.19"S	Small	12/1/2002	None	671
	16°11'20.00"E		*16*		*10*
North 53	28°24'12.04"S	Intermediate	12/1/2002	None	2435
	16°10'56.16"E		*16*		*16*
North 54	28°23'57.51"S	Small	12/1/2002	None	100
	16°10'46.98"E		*16*		*1*
North 55	28°24'1.60"S	Small	12/1/2002	None	165
	16°10'41.89"E		*16*		*1*
North 56	28°23'52.82"S	Big	12/1/2002	None	0
	16°10'41.77"E		*16*		*0*
North 57	28°23'44.81"S	Intermediate	12/1/2002	None	1432
	16°10'27.02"E		*16*		*10*
North 58	28°23'29.33"S	Small	12/1/2002	None	1312
	16°10'6.23"E		*16*		*6*
North 59	28°23'22.44"S	Small	12/1/2002	None	1369
	16°10'0.32"E		*16*		*7*
North 60	28°23'16.05"S	Small	12/1/2002	None	173
	16° 9'58.21"E		*16*		*5*
North 61	28°23'12.10"S	Small	12/1/2002	None	5
	16° 9'55.88"E		*16*		*2*
North 62	28°23'8.55"S	Intermediate	12/1/2002	None	1148
	16° 9'44.61"E		*16*		*13*
North 63	28°22'55.55"S	Small	12/1/2002	None	621
	16° 9'32.64"E		*16*		*4*
North 64	28°22'48.91"S	Small	12/1/2002	None	571
	16° 9'28.33"E		*16*		*7*
North 65	28°22'38.12"S	Small	12/1/2002	None	1820
	16° 9'15.42"E		*16*		*8*
North 66	28°22'28.77"S	Small	12/1/2002	None	417
	16° 9'2.68"E		*16*		*7*
North 67	28°22'17.21"S	Small	12/1/2002	None	0
	16° 9'1.21"E		*16*		*0*
North 68	28°22'22.36"S	Small	12/1/2002	None	0
	16° 8'51.95"E		*16*		*0*
North 69	28°22'14.93"S	Small	12/1/2002	None	0
	16° 8'48.40"E		*16*		*0*
North 70	28°22'8.56"S	Small	12/1/2002	None	820
	16° 8'40.09"E		*16*		*9*
North 71	28°22'1.96"S	Small	12/1/2002	None	0
	16° 8'31.39"E		*16*		*0*
North 72	28°21'56.95"S	Small	12/1/2002	None	139
	16° 8'28.21"E		*16*		*4*
North 73	28°21'55.27"S	Small	12/1/2002	None	18

	16° 8'24.18"E		*16*		*1*
North 74	28°21'49.43"S	Intermediate	12/1/2002	None	3522
	16° 8'13.52"E		*16*		*11*
North 75	28°21'42.84"S	Small	12/1/2002	None	1631
	16° 8'7.91"E		*16*		*11*
North 76	28°21'38.43"S	Small	12/1/2002	None	152
	16° 8'2.32"E		*16*		*2*
North 77	28°21'33.94"S	Small	12/1/2002	None	803
	16° 7'56.58"E		*16*		*10*
North 78	28°21'22.46"S	Small	12/1/2002	None	0
	16° 7'43.60"E		*16*		*0*
North 79	28°21'16.51"S	Small	12/1/2002	None	196
	16° 7'35.81"E		*16*		*2*
North 80	28°21'8.98"S	Intermediate	12/1/2001	None	150
	16° 7'23.57"E		*16*		*1*
North 81	28°21'1.57"S	Small	12/1/2001	None	0
	16° 7'14.76"E		*17*		*0*
North 82	28°20'56.13"S	Small	12/1/2001	None	175
	16° 7'9.51"E		*17*		*5*
North 83	28°20'48.03"S	Small	12/1/2001	None	430
	16° 6'59.06"E		*17*		*8*
North 84	28°20'40.08"S	Small	12/1/2001	None	4186
	16° 6'51.85"E		*17*		*12*
North 85	28°20'32.30"S	Small	12/1/2001	None	184
	16° 6'44.99"E		*17*		*4*
North 96	28°17'38.84"S	Small	12/1/2001	None	15
	16° 3'53.85"E		*17*		*1*
North 97	28°17'29.90"S	Small	12/1/2001	None	268
	16° 3'42.02"E		*17*		*4*
North 98	28°17'24.42"S	Small	12/1/2001	None	336
	16° 3'35.85"E		*17*		*4*
North 99	28°17'16.39"S	Small	12/1/2001	None	365
	16° 3'26.30"E		*17*		*1*
North 100	28°17'11.99"S	Small	12/1/2002	None	591
	16° 3'21.60"E		*16*		*13*
North 101	28°17'6.56"S	Small	12/1/2001	None	30
	16° 3'18.20"E		*17*		*1*
North 102	28°17'2.96"S	Small	12/1/2001	None	1240
	16° 3'11.40"E		*17*		*6*
North 131	28°12'56.68"S	Intermediate	12/1/2001	None	393
	15°58'2.35"E		*17*		*9*
North 132	28°12'51.73"S	Small	12/1/2002	None	134
	15°57'54.77"E		*16*		*4*

APPENDIX 3.2:
SIMPER analysis of species for North, Middle and South, listing their individual contributions and cumulative contributions to each of the three areas, listed in order of importance. Mean abundances are numbers per pond summed over eight counts.

Taxon	Contribution (%)	Cumulative (%)	NORTH Mean abundance	MIDDLE Mean abundance	SOUTH Mean abundance
Blacknecked Grebe	29.49	29.49	329	8.93	7.42
Cape Cormorant	12.46	41.95	19.8	11.2	230
Greater Flamingo	10.63	52.58	47.3	114	37.5
Kelp Gull	9.61	62.19	20.8	37.9	64.8
Common Tern	6.07	68.26	65.4	25.3	60.2
Lesser Flamingo	5.16	73.42	6.51	20.9	37.5
Pied Avocet	3.98	77.41	30.3	4.27	5.71
Swift Tern	3.84	81.25	38.3	16.3	10.8
Whitebreasted Cormorant	3.81	85.06	21.8	2	17.3
Hartlaub's Gull	3.19	88.25	6.64	9.13	22.5
Cape Teal	3.00	91.25	24.9	11.1	5.96
Sanderling	1.75	92.99	13.4	4.33	9.33
African Spoonbill	1.54	94.54	0	0	6.21
Blackwinged Stilt	1.42	95.95	5.44	1.6	1.92
Sandwich Tern	1.26	97.21	3.24	6.93	2.88
South African Shelduck	0.56	97.78	2.62	0.27	2.08
Grey Heron	0.49	98.27	0.02	0.07	5.33
Kittlitz's Plover	0.43	98.70	1.55	1.47	3.42
Bank Cormorant	0.38	99.07	1.25	0.13	2.25
Whitefronted Plover	0.25	99.32	0.67	0.33	1.67
African Black Oystercatcher	0.13	99.46	0	0	3.63
Common Greenshank	0.11	99.56	0.14	0.07	0.13
Sacred Ibis	0.09	99.66	0	0.40	0.83
Little Egret	0.08	99.74	0	0	1.75
Egyptian Goose	0.07	99.81	0	0	0.87
Greyheaded Gull	0.06	99.88	0	1.07	0.29
African Black Duck	0.03	99.91	0	0.20	0.08
Pied Kingfisher	0.03	99.94	0.04	0	0.75
Little Grebe	0.02	99.95	0	0	0.08
Blacksmith Plover	0.01	99.97	0	0	0.29
Common Whimbrel	0.01	99.98	0	0	0.12
Goliath Heron	0.01	99.98	0	0	0.08
Common Ringed Plover	0.01	99.99	0.02	0	0.08
Threebanded Plover	0.004	99.99	0	0	0.04
Curlew Sandpiper	0.004	100	0.11	0	0
Marsh Sandpiper	0.003	100	0	0	0.04

APPENDIX 3.3:

List of surveys from which information on bird numbers was drawn to make comparisons with the mining ponds. Bold values for shore length or area of site were taken from the reference cited; other values were based on this value or estimated.

Site	References	Dates of surveys	Length of site surveyed km	Area of site km^2	Mean/Median/total no wetland birds	Minimum	Maximum	Mean Density km^{-1}	Density km^{-2}	Number of wetland spp	Regionally threatened spp	Species >1% southern African population
Baia dos Tigres, Angola	Wearne and Underhill 2005	Unspecified	**64**		2050			32	N/A	25		
Orange River	Williams 1986	Dec-85	20	22.98	23653			1183	51	53	14	9
Orange River	Williams 1986	Apr-86	20	22.98	7562			378	16	50		
Orange River	Ryan & Cooper 1985	Jan-80	**9.8**	22.98	21611			2205	96	56		
Orange River	Anderson et al. 2003	Apr 1996 to Dec 2001	20	18	5909	2510	9240	295	16	54		15
Orange River	Anderson et al. 2003	Jan 1980 - Dec 1995	9.8	18	12653		26653	1291	703	54		
Orange River	Simmons 1994	Jan-94	9.8	**22.98**	15281			1559	665	46		
Orange River	Simmons 1995	Apr-94	9.8	22.98	1509			154	66	41		
Orange River	Simmons 1995	Jan-95	9.8	22.98	7538			769	328	43		
Orange River	Kolberg 2017	Winter 2016	9.8	22.98	886			90	39	26		
Orange River	cwac.adu.org.za	Summer & winter 1997	9.8	22.98	824	379	25212	84	36	74		
Orange River mouth overall					9743	379	26653	801	202	26-72	14	15
Southern Angola coast	Simmons et al. 2006	Dec 1998, 2001	**205**		6677			**33**	N/A	25	7	
N Cape coast	Ryan and Cooper 1985	jan-80, jan-82	**172**	N/A	61378			892	N/A	63		
Skeleton Coast (Kunene to Cape Cross)	Ryan and Cooper 1984	Nov-Dec 1981	**574**	N/A	37549			65	N/A	29		
Durissa Bay to Sandwich Harbour	Whitelaw et al. 1978	Dec 1976 - Jan 1977	**225**	N/A	44921			200	N/A			
Swakop River to	Whitelaw et al. 1978;	Dec 1976 -	**30**	N/A	12435			449	N/A	32		

Walvis Bay	Simmons et al. 1999	Jan 1977										
Luderitz: Grossebucht to Agate Beach	Hockey 1982	Mar-81	**46**	N/A	2659			58	N/A			
30-km beach	Simmons et al. 1999	Not specified	30	N/A	13000			433	N/A	13		
Open coast overall					25517	2659	37549	304		13-63	7	
Luderitz Lagoon	Noli-Peard and Williams 1991		17	**5**	7270			428	1454	24		6
Sandwich Harbour	Whitelaw et al. 1978	Dec 1976 - Jan 1977	15	20	60050			4003	3003			
Sandwich Harbour	Simmons et al 1999	Various	15	20	35000			2333	1750	87	4	14
Sandwich Harbour	Noli-Peard and Williams 1991	Unspecified	15	20	16500	11000	22000	1100	825	67		
Sandwich Harbour	Kolberg 2015	1990-2015 summer	20	30	137500	38000	402000	6875	4583			14
Sandwich Harbour sections	Mendelsohn et al. 2002	18-31 counts	20	30	75521	5495	333368	3776	2517	40		
Sandwich Harbour	Kolberg 2015	1990-2015 Winter	20	30	58800	25000	148000	2940	1960	85		
Sandwich Harbour	Kolberg 2017	Winter 2016	20	30	100066			5003	3336	29		
Sandwich Harbour overall					69062	5495	402000	3719	2568	29-87	4	14
Walvis Bay	Simmons et al. 1999	Various	15	**594**	75000	50000		5000	149		3	16
Walvis Bay	Whitelaw et al. 1978	Dec 1976 - Jan 1977	15	504	49351			3290	98	35		
Walvis Bay Lagoon & salt works	Mendelsohn et al. 2002	27 counts	15	504	70418	5891	155765	4695	140			
Walvis Bay	Williams 1987	Aug-87	15	504	75039			5003	149	32	2	2
Walvis Bay	Noli-Peard and Williams 1991	Various	15	504	59179	37500	150000	3945	117		11	6+
Walvis Bay	Underhill and Whitelaw 1977	Jan-77	15	504	47998			3200	95	28		
Walvis Bay	Wearne and Underhill 2005	Jan 1997-2005	15	**504**	148739		401313	9916	295	47		
Walvis Bay	Wearne and Underhill 2005	July 1997-2005	15	**504**	80592		163677	5373	160	45		
Walvis Bay overall					75790	5891	401313	5053	150	28-47	11	16

Mile 4 Salt works	Simmons et al. 1999	Various	20	34	95000			4750	2794	28	3	3
Cape Cross lagoon	Simmons et al. 1999 overview	Unspecified	15	23	11000		72200	733	478		3	5
Cape Cross lagoon	Kolberg 2017	Winter 2016	15	23	37945			2530	1650	20		
Cape Cross lagoon overall					24473	11000	72200	1632	1064		3	5
Cunene Mouth Lagoon	Simmons et al. 1993	Mar-May 1991	**2.36**	**1.25**	4204			1781	3363	55	14	6
Cunene Mouth Lagoon	Simmons et al. 1993	Nov-91	**2.36**	**1.25**	4966			2104	3973	45		8
Cunene Mouth Lagoon	Braine 1990	1982-1988	2.36	1.25	6543			2772	5234	60		
Cunene Mouth Lagoon	Ryan and Cooper 1984	Nov-Dec 1981	2.36	1.25	2074			879	1659	45		
Cunene Mouth Lagoon	Noli-Peard and Williams 1991	Unspecified	4	1.25	2440			610	1952	80		
Cunene Mouth Lagoon	Simmons et al. 1998b, 1999	Unspecified	2.36	1.25	869		3900	217	695	72	12	
Cunene mouth	Anderson et al. 2001	Jan-01	4.2	1.8	2297			547	1276	41		
Kunene	Mendelsohn et al. 2002		4.2	1.8	611	19	1757	145	339	46		
Cunene Mouth overall					3001	611	6543	1132	2312	41-80		
Diamond Mine Ponds	Chapter 3	2018, 8 surveys over a year	**75**	**11.18**	11028			147	986	36	6	9

CHAPTER 4: Fish Associated with the Mining Ponds

INTRODUCTION

A great diversity of fish utilizes surf zone habitats and estuaries along the Southern African coast as nursery and feeding areas (Lasiak 1981; Whitfield 2019), where early growth and development take place during the life cycles of estuarine-dependent marine species (Potter et al. 2015; Whitfield 2019). The potential of a site as a nursery area depends on food availability and shelter from predators, as well as its accessibility to the fish. I do, however, recognise that for a nursery function to be fulfilled, the life cycle of marine species must be completed by returning to sea to breed, and there is the danger that the ponds may become 'coastal traps' if they prevent return to the sea.

In this chapter, I investigate the ecological importance of the mining ponds generated by coastal diamond mining in southern Namibia, in terms of their occupation by fish, as the ponds offer a sheltered and nutrient-enriched environment than can be used by both adult fish, and juveniles occupying them as nursery areas (Whitfield 1994). It is postulated that fish larvae or juveniles enter the mining ponds during breaching occurrences or when seawater overtops the seawalls during heavy seas. Fish larvae can also be transferred between ponds when water is pumped from one pond to another or the ponds become physically connected.

The study site lies north of the Orange River mouth, the longest river in southern Africa and one of few perennial rivers in the region that flow into the Atlantic Ocean (Simmons and Allan 2002). Bethune and Roberts (1991) consider that the Orange River fish community is less diverse than other permanently flowing rivers in Namibia, including the Kunene, Kavango, Zambezi and Chobe rivers. Lamberth (2013) compared the fish assemblage of the Orange River Estuary with those of other estuaries found along the west coast of southern Africa in which both freshwater species and freshwater-tolerant estuarine-dependent marine species are found, and listed 36 fish species representing 19 families for the Orange River, with eight being marine species.

Namibia's coast hosts two biogeographic zones lying respectively northwards and southwards of Lüderitz, with both regions regarded as cool-temperate (Emanuel et al. 1992; Harrison and Whitfield 2006; Potts et al. 2015), but with the northern section being impoverished in terms of species richness (Emanuel et al. 1992; Engledow and Bolton 1994, 2003). My study site falls within the southern region of the Namibian coastline, where there is a scarcity of estuaries (Potts et al. 2015) and fish species richness and abundance are low when compared with warm temperate and subtropical systems (Harrison and Whitfield 2006).

The Southern Coastal Mines are flanked by the nutrient-rich Benguela Ecosystem, which supports important fish populations that form the basis for the Namibian fishing industry (O'Toole 1997; Hutchings et al. 2009). Mining operations along the south-western Namibian coastline have resulted in the creation of multiple coastal seawater ponds that generate a unique habitat with the potential to be occupied by fish. My general hypothesis is that the ponds will support and provide habitat for marine fish species found along the south-western coastline of southern Africa.

More specifically, I address the following issues:

1. I quantified the diversity of fish fauna occupying selected ponds, including identification of the species present and an assessment of their relative abundance, and hypothesised that (a) there would be significant differences in community composition among areas in the north, middle and south, because of the older age and higher salinities of ponds in the north, and (b) the diversity and relative amounts of fish would be less in the north ponds, which are warmer, older and more saline (based on information in Chapter 1).
2. I measured the size composition and body condition of the two dominant species in the ponds, with the expectation that (a) mean sizes would be greatest in the north ponds because of their greater age, (b) recruits would be absent from north ponds because of the prolonged isolation of these ponds from the sea, and (c) body condition would be lowest in the north ponds for steenbras due to food shortage and salinity stress, and highest for mullet in middle ponds because high chlorophyll-*a* levels there may fuel particulate and detrital food for mullet.

3. I determined diets by assessing stomach contents and calculated Gut Fullness Indices (GFIs) to test the hypotheses that steenbras would have the smallest range of diet and the lowest GFIs in the north where benthic invertebrate food is less diverse and less abundant (J. Cloete, pers. comm. and personal observations) and salinities reach stressful levels and that mullet would have the lowest GFIs in the north due to salinity stress.
4. I compared the gonadosomatic indices (GSIs) of mullet and steenbras among north, middle and south areas, and assessed the seasonality of GSIs of steenbras, anticipating that GSI would be lower in the north than middle or south ponds.
5. Growth rates and ages of the two dominant species found in the ponds were determined by examining otoliths of individuals, and were compared to values obtained from the literature on studies of fish caught from the ocean. As there were no *a priori* reasons to expect growth would be higher or lower in the ponds than in the sea, the null hypothesis was to predict no difference.

MATERIALS AND METHODS

Study area

The sites I examined lie within the Southern Coastal Mines, Namdeb's southernmost mining license area, which extends approximately one hundred kilometres north from the Orange River mouth and contains multiple marine ponds created by mining activities.

The distance covered by these mining ponds is approximately 75 kilometres, with 150 ponds in total stretching along the coast. The study site was divided into three areas namely north, middle and south, based on differences in the post-mining ages of the ponds (as outlined in Chapter 1). The north contained 111 ponds, the middle 15 ponds and the south 24 ponds.

Most of the study focused on the two dominant species, the west coast steenbras *Lithognathus aureti* (henceforth called 'steenbras') and the southern mullet *Chelon richardsonii* ('mullet' – previously named *Liza richardsonii*), but any other species encountered were also recorded. Only those species captured during systematic sampling were included in quantitative analyses, but any other species encountered incidentally were recorded.

Sampling method

Intense sampling was carried out in five selected ponds per area during December 2017 to January 2019 (Table 4.1). Physical conditions in these ponds (taken from Chapter 1) appear in Appendix 4.1 (p 170). None of the physical variables differed among areas in a consistent manner except for chlorophyll-a, which appeared higher in the middle ponds, and salinity, which was high in the north (70-88‰) but close to the value of seawater in the south and middle ponds (35-41‰). As a result, I explored only the relationships between fish numbers and these two variables.

The fish fauna of the mining ponds was sampled using a mining dinghy and mesh seine net (50 m x 4 m with a 19-mm mesh). The number of net pulls per pond varied from one to three, and was determined by the need to sample at least three mullet and three steenbras per pond per sampling date for comparative analyses. The total number of net pulls differed by no more that 5% among the three areas. Sampling was carried out during daylight hours mostly between 8h00 and 15h00.

Each pond was sampled once-off except for two ponds, S3 and N49, where an additional three fish from each pond were caught monthly with a casting rod from April 2018 to September 2018. Sample size was limited by permit conditions imposed on anglers. I pooled the data to assess the gonad cycles of steenbras in the ponds over time.

Otoliths were extracted from steenbras and mullet to determine the growth rates and ages of fish among areas. Additional otoliths of steenbras caught from the ocean, using casting rods, were also used for comparisons, along with available literature on this fish's growth performance in the sea; but as the sample of sea-caught fish was small, I used information from them in a qualitative manner only.

The fish were living in mining ponds that were being 'dewatered' as part of the mining (Chapter 1). Once the water levels in these ponds dropped, the fish were caught using nets as part of the mariculture venture conducted in the mining area. These fish would therefore have died whether or not they were sampled for scientific purposes.

Most specimens were supplied freshly dead to me by the aquaculture team. Supplementary collections of live fish were caught from S3 and N49 ponds not sampled by the aquaculture venture. To conform with ethical requirements specified by the University of Cape Town's Science Faculty Ethics Committee and their permit conditions (permit 2018/v9/GMB), these fish were subdued by application of 0.1% concentrated clove oil as advocated by Fernandes et al. (2017), which calms and pacifies them, before the spinal cord was severed behind the head.

Fish were measured, weighed, dissected and their gonad development assessed following Albieri and Araújo's (2010) classification: immature (threadlike), developing (thicker and ribbon-like), mature (filling almost 2/3 of the abdominal cavity), ripe or ripe running (the two being defined as the same thing by Albieri and Araújo: occupying the entire abdominal cavity), or spent/recovering (wrinkled, occupying ~1/3 of the abdominal cavity). Other parameters recorded during sampling included (a) species composition, abundance, individual masses and sizes (fork lengths) of fish; (b) stomach contents (composition and weight of full and empty stomach) for an assessment of diet and gut fullness; (c) extraction, weighing and stage-classification of gonads to determine reproductive state; (d) extraction of otoliths for determination of age and growth rates, by counting the annual rings on otoliths, following the methods described by Griffiths et al., (2002).

Gut fullness was determined by excising the stomach, weighing it, emptying the stomach contents and then weighing the empty stomach, and then applying a Gut Fullness Index (see below in Statistical Analysis). The stomach contents were then spread uniformly across a petri dish and the identity, number and estimated volumes of each food item assessed visually.

Some fish were caught in the sea (six steenbras and one silver kob *Argyrosomus inodorus*), but because of the small sample size, only the steenbras were employed to obtain measures of condition index and growth rate. Sample sizes for univariate metrics are indicated on the data in the Results, but for each species, spanned 3-41 per month per area. To assess the age composition of steenbras and mullet in each of the north, middle

and south ponds, the size-frequency distributions in each area were converted to ages, estimated from the growth curves for each species.

Table 4.1: Fish sampling ponds

Pond No.	Sampling Date	South GPS Coordinates	East GPS Coordinates
South 3	5/1/2018	28°35'42.82"S	16°23'43.95"E
South 16	19/4/2018	28°33'3.49"S	16°19'54.86"E
South 17	27/4/2018	28°32'20.71"S	16°19'14.75"E
South 20	24/5/2018	28°31'43.16"S	16°18'28.90"E
South 23	1/6/2018	28°31'8.45"S	16°18'7.49"E
Middle 25	24/5/2018	28°30'33.31"S	16°17'24.77"E
Middle 26	24/5/2018	28°30'4.85"S	16°16'55.56"E
Middle 27	28/12/2017	28°30'35.38"S	16°17'19.53"E
Middle 28	19/12/2017	28°30'29.57"S	16°16'56.42"E
Middle 39	4/1/2019	28°27'13.34"S	16°13'56.06"E
North 44	12/12/2017	28°26'2.39"S	16°12'55.17"E
North 49	9/1/2018	28°24'43.01"S	16°11'33.66"E
North 102	22/12/2017	28°17'2.96"S	16° 3'11.40"E
North 109	23/12/2017	28°15'55.87"S	16° 1'55.48"E
North 116	2/1/2018	28°15'15.40"S	16° 1'1.16"E
Ocean	5/1/2019	N/A	N/A

Statistical analysis

The statistical software JMP (version 14) was used to analyse species abundance, length, weight, sex, gonadal stages and age per area and per pond (in nested analyses). Scatterplots were employed to display species log length versus log weight, and full stomach weights versus empty stomach weights – per area and per pond.

A second statistical program PAST (Paleontological Statistics) version 2.17c was used to analyse species composition per pond in each area. PERMANOVA determined the significance of differences in community composition among areas, cluster analysis was utilised to display patterns of similarity among ponds, and SIMPER to assess species' contributions to similarity and dissimilarity among ponds. Diversity was explored by comparing numbers of species (species richness), the Shannon Index of alpha diversity and the Shannon evenness index among the three areas (north, middle, south).

Abundances of each species in each area were the summed counts per area (five ponds per area) and data were not transformed for statistical analysis for PERMANOVA or Cluster analyses.

Size compositions of both steenbras and mullet were compared with one-way ANOVA, with area (south, middle, north) as a fixed factor. Data were log-transformed because of inequalities of variance in the untransformed data. Body condition was determined by plotting log mass against log length for the total set of data for each species in each area. Regressions were also derived for each area separately to graphically illustrate differences. Body condition in each area was quantified as the magnitude of positive or negative residual of each fish from the regression for the total population, and ANOVA used to test for differences in residuals among areas.

Gut fullness was assessed as a Gut Fullness Index, GFI (Haley et al. 2011):

$$\text{GFI} = \text{Mass of stomach contents} / \text{Mass of empty stomach} \times 100$$

Differences in GFI among areas were examined with ANOVA, with areas as a fixed factor. As there was no significant difference between the GFIs of females, males and protandrous individuals (ANOVA $F_{2,115} = 1.66$, $p = 0.19$), the three types of sexual state were pooled when analysing GFIs.

For both mullet and steenbras, a gonadosomatic index GSI (Wuenschel et al. 2018) was derived for each individual fish from north, middle and south ponds:

$$\text{GSI} = \text{Gonad mass} / (\text{body mass-gonad mass}) \times 100$$

Data for both sexes were combined in analyses to increase sample size. I recognise this could have introduced bias if samples were skewed towards either sex, but as the GSI was standardised relative to body length, this would have offset any bias. For steenbras, the significance of the difference between the GSI of middle and south ponds combined vs. north ponds was determined with a Student's T-test. In addition, a two-way ANOVA was used to determine the effects of the fixed factors area and month on the GSI. For this latter analysis, the data set was reduced to a sample size of three per month and per area to establish a balanced design, by using a random table selection of data points that were

omitted. Ten simulations were run with different random selections of the data omitted to achieve this, but as all yielded the same statistical outcome, only a single run is presented. For mullet, fish were not captured in all months, so the effect of months could not be assessed: instead, the data were pooled across the months for which data existed (December 2017 to May 2018) and a one-way ANOVA run to compare among North, Middle and South ponds. As the design was unbalanced, with different numbers of samples in the three areas, two tests were run. In the first, the total data set was employed, whereas in the second, the data were reduced to a constant sample size of 12 per area by first removing outliers, and then random omission of data, with the stipulation that the mean values of the random sample remained within 5% of the means of the total data set. As the two tests yielded closely similar outcomes and the balanced design was more robust, I elected to use the analysis of the reduced data set while presenting the full data set graphically.

For growth rates, lengths of individual fish that had been aged from otolith analyses were plotted against age, and based on Ford-Walford plots, von Bertalanffy growth curves were derived for steenbras and for mullet, with t0 assumed to be 0. As sample sizes were small, the data were pooled for all ponds for analysis and across all stages of sexual status, although it is recognised that males and females may grow at different rates (Horton et al. 2019). Data points for all three areas are, however, distinguished on the growth-rate graphs to allow qualitative comparisons.

RESULTS

Species abundance

The marine ponds were occupied by west coast steenbras *Lithognathus aureti* ('steenbras'), southern mullet *Chelon richardsonii* ('mullet') and to a lesser extent white stumpnose *Rhabdosargus globiceps* ('stumpnose') and silver kob *Argyrosomus inodorus* ('kob'). The total number of fish caught was 332 from the 15 surveyed ponds (5 ponds per area) (Fig. 4.1).

The north area had the most fish (summed count 150) consisting of 57.3% southern mullet and 42.7% west coast steenbras, but no other species were recorded there. The middle area had the least fish (summed count 72) with 58.3% southern mullet and 41.7%

west coast steenbras. The south area had a summed count of 110, of which 38.2% were southern mullet, 55.5% west coast steenbras, 5.4% white stumpnose and 0.9% kob (Fig. 4.1). Although not captured in dedicated surveys, isolated individuals of pipefish *Syngnathus temminckii*), small unidentified gobiid and clinid species, and the serpent eel *Ophisurus serpens* were recorded in south ponds, but never in middle or north ponds. Six west coast steenbras and one kob were caught in the ocean.

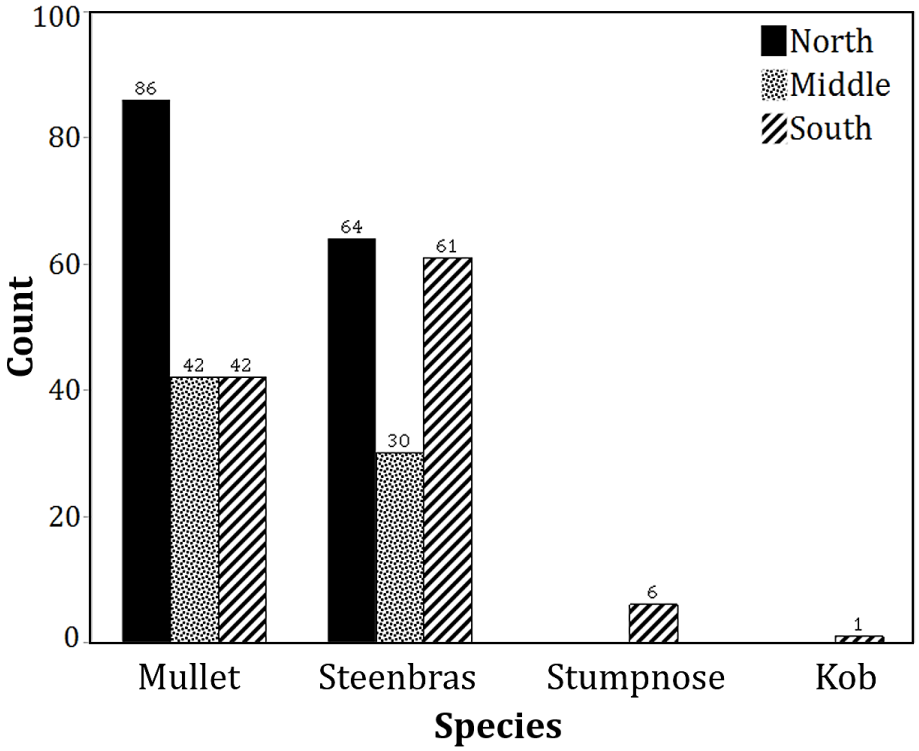

Figure 4.1: Fish species per area (summed counts, n = 332)

Physical variables and relationships with fish abundance

Of the physical variables (Appendix 4.1), pH, chl-*a* and dissolved oxygen were not significantly different among areas (ANOVA, $df_{2,12}$, $p>0.133$ in all cases). Temperature did differ (ANOVA, $F_{2,12} = 10.72$, $p = 0.002$). Differences in salinity were even more marked (ANOVA, $F_{2,12} = 113.46$, $p < 0.0001$), averaging 78.7‰ in the north ponds, double the mean values of 38.7 and 37.7‰ in the south and middle ponds. As I regarded salinity as the critical factor likely to influence fish assemblages, I explored its relationship with fish numbers. Surprisingly, salinity was not significantly linearly correlated with total fish numbers ($r = 0.178$, $df = 13$, $p>0.3$) or with total numbers of mullet ($r = 0.436$, $df = 13$, $p = 0.133$). For steenbras, the correlation was also non-significant ($r = 0.454$, $df = 13$, $p =$

0.106), but a negative exponential relationship was significant ($Y = 21.503e^{-0.036X}$; $R^2 = 0.336$, $p = 0.025$).

Species composition and pond similarity

There were no significant differences in species composition among the North, Middle and South, based on species composition and relative abundance (PERMANOVA, $F_{2,12} = 0.9034$, $p = 0.4693$), and post-hoc pair-wise comparisons confirmed that none of the areas differed significantly (Table 4.2).

Table 4.2: One-Way PERMANOVA for North, Middle and South. Values are probabilities of differences among areas

	SOUTH	MIDDLE	NORTH
SOUTH	0	0.6636	0.2097
MIDDLE	0.6636	0	0.6549
NORTH	0.2097	0.6549	0

Of the four fish species recorded in dedicated samples, only two (mullet and steenbras) were present in the north and middle ponds, whereas four species occurred in the South, with the addition of stumpnose and kob (Table 4.3). Measures of diversity (species richness and Shannon Index) were greatest in the South, although evenness peaked in the North and Middle.

Table 4.3: Diversity indices for North, Middle and South (excluding fish caught in the Ocean), Pooled data per area

	North	Middle	South
No. of species	2	2	4
Shannon	0.682	0.679	0.896
Evenness	0.9893	0.9861	0.6124

Despite the absence of any significant difference among areas in the PERMANOVA, I used Similarity Percentage (SIMPER) to identify the main species characterising north, middle and south ponds (Table 4.4). Southern mullet and west coast steenbras were cumulatively responsible for 95.97% of the differences among catches in North, Middle and South. The north accounted for the greatest proportion of southern mullet and west coast steenbras. The remaining two species were found in small numbers and in the south ponds only.

Southern mullet were most abundant in the north ponds (86 of the 150 – N49:16, N102:31, N109:9 and N116:30). The majority of west coast steenbras in north ponds were caught from N49 (63), plus one individual in N44 (Fig. 4.2); but is was striking that mullet were the sole occupants of the other three north ponds. The middle ponds were also most abundant in southern mullet (42 of 72 – M25:17, M27:20 and M28:5). West coast steenbras were found in M26:2, M28:25 and M39:3 (Fig. 4.2).

West coast steenbras were present in all five south ponds and accounted for over half of the counts of fish caught in the area. The remaining fish were 46 mullet caught from S13 and S16, six white stumpnose in S16 and one silver kob in S23 (Fig. 4.2).

Table 4.4: SIMPER analysis of fish species for north, middle and south ponds, listing their individual contributions and cumulative contributions to each of the three areas. Mean abundances are numbers per pond summed.

Taxon	Contribution (%)	Cumulative (%)	NORTH Mean abundance	MIDDLE Mean abundance	SOUTH Mean abundance
Mullet	51.9	51.9	17.2	8.4	8.4
West Coast Steenbras	44.07	95.97	12.8	6	12.2
White Stumpnose	2.354	98.32	0	0	1.2
Silver Kob	1.678	100	0	0	0.2

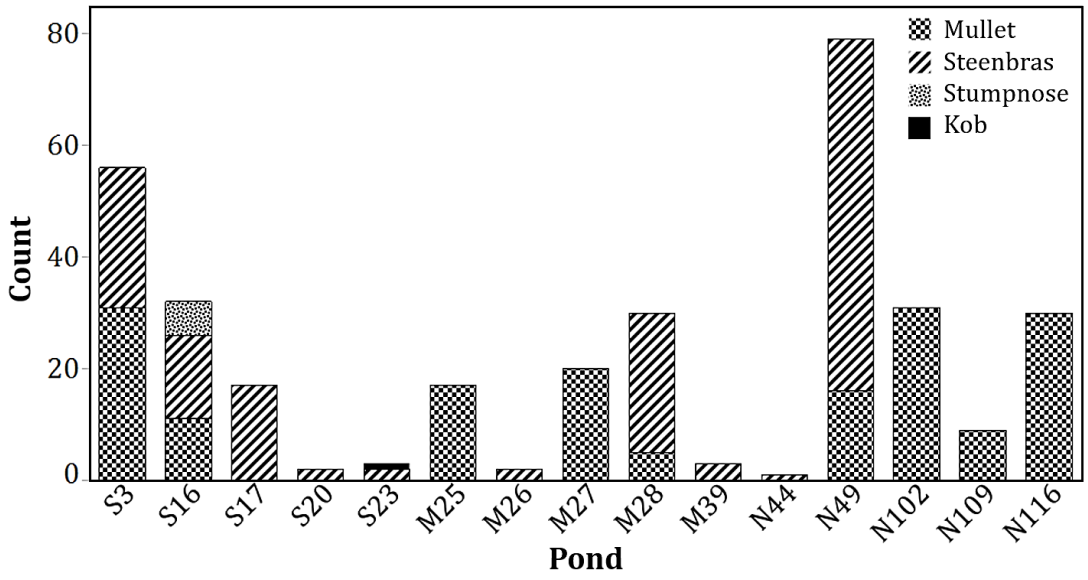

Figure 4.2: Species composition per pond in north (n = 150), middle (n = 72) and south ponds (n = 110)

The hierarchical clustering dendrogram of community composition, based on the similarity amongst fish species per pond using Ward's similarity linkage (Fig. 4.3), reflected the non-significant differences among areas detected in the PERMANOVA analysis: there was no obvious clustering of ponds related to differences among the three areas (north, middle and south).

Five major clusters emerged from the Cluster analysis:

a) Cluster 1 comprised a single pond (N49) containing high counts of both steenbras (63) and mullet (16): the only north pond with both species.
b) Cluster 2 covered two middle (M25, M27) and two north (N102, N116) ponds where only mullet were present.
c) Cluster 3 had two sub-clusters, overall containing two ponds from each area, all with low counts of fish. Sub-cluster 3a consists of M39, S20 and S23 housing steenbras only. A breach event occurred at M39 on September 2018 (see Fig. 1.6, Chapter 1) and flooded that mine area, creating a pond and trapping fish that entered from the sea. Sub-cluster 3b comprised ponds M26 and N44, which also had steenbras only, and N109, with mullet only.
d) Cluster 4 comprised a single pond (S3-mariculture pond) containing high counts of both steenbras (25) and mullet (31).

e) Cluster 5 comprised two southern dewatering ponds (S16 and S17) and a middle pond (M28) that had high numbers of steenbras; S16 and M28 also had mullet present.

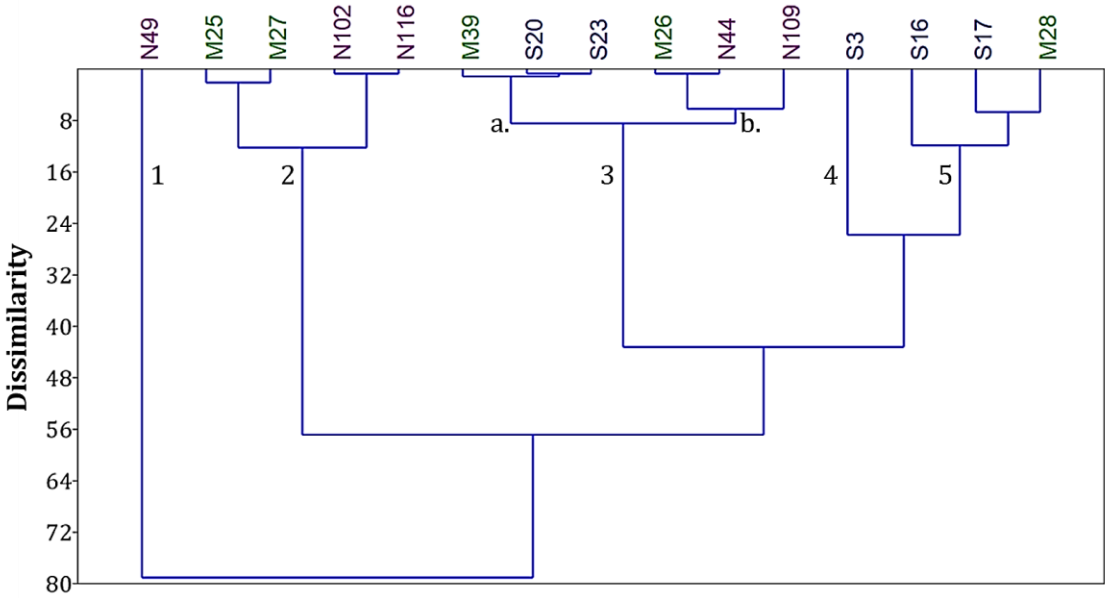

Figure 4.3: Dendrogram representing the similarity amongst fish species per pond

Fish lengths and body condition

This section focuses on the sizes and condition of the southern mullet *Chelon richardsonii* and the west coast steenbras *Lithognathus aureti* (identified in Table 4.4 as the two dominant contributors in the ponds).

The size compositions of the two species (Fig. 4.4) revealed two patterns. First, there were significant differences among areas in mean sizes (log-transformed for analysis) for both steenbras ($F_{2,152} = 31.18$, $p < 0.0001$) and mullet ($F_{2,167} = 33.58$, $p < 0.0001$). Second, mean sizes were influenced largely by the presence or absence of a recruit cohort (0-15 cm) of steenbras in the south ponds and of mullet in both south and middle ponds. In contrast, recruits of both species were absent from north ponds. Post-hoc Tukey tests showed that for mullet all three areas differed significantly from each other, but for steenbras only south ponds differed from north and middle ponds ($p < 0.05$) (Fig. 4.4).

Steenbras reached mean sizes in the north, middle and south ponds of respectively 44.25, 55.18 and 32.56 cm; and corresponding mean masses of 981, 2948 and 906 g (Table 4.5).

For mullet, the respective mean lengths were 40.5, 37.9 and 14.3 cm; and the mean masses 619, 833 and 167 g (Table 4.5).

Although large mullet were found in the north, peaking in N102 (mean length 46.2 cm and mean mass 847 g), the mullet caught from pond M27 in the middle area (mean length 48.5 cm and mean mass 1082 g) were the largest. Recruits (< 10cm) were found only in S3 and M25. Mullet caught in the north were all > 20 cm (Fig. 4.4b). Recruits of steenbras (< 13cm) were found only in S17.

Body condition ('health') of fish was assessed by plotting log-log length-weight regressions for populations in the north, middle and south ponds, and comparing actual masses of fish of a given size with those predicted from a log-log regression for all individuals combined. Positive or negative deviations from predicted values (i.e., residuals) were taken as measures of, respectively, the 'good' or 'poor' body condition of the fish (Fig. 4.5).

ANOVAs of these residuals indicated that there were highly significant differences in body condition among north, middle and south areas for both steenbras ($F_{3,139} = 61.12$, $p < 0.0001$) and mullet ($F_{2,167} = 40.69$, $p < 0.0001$). Post-hoc Tukey tests showed that for mullet all three areas differed significantly from each other, but for steenbras significant differences were confined to the ocean and south ponds versus north and middle ponds ($p < 0.05$) (Fig. 4.6).

For steenbras the mean (log10) values for condition index were -0.1135 for north ponds, 0.0505 for middle ponds, 0.1088 for south ponds, and 0.1769 for samples collected in the sea. The ocean thus had the 'healthiest' fish, and north ponds decisively had the fish with the poorest condition. For mullet, condition index was lowest in south ponds (-0.1792), close to average in north ponds (0.0019) and highest in middle ponds (0.1751). As I suspected that body condition in the three areas might be influenced by chlorophyll-*a* concentrations in the ponds (Appendix 4.1), I tested to see if the higher chlorophyll-*a* levels in the middle ponds (averaging 24.8 $\mu g.L^{-1}$) were significantly different from those in the south (10.3 $\mu g.L^{-1}$) and the north (8.6 $\mu g.L^{-1}$). Because of high variance, however, the difference was not significant (ANOVA $F_{2,12} = 1.35$, $p = 0.294$).

Table 4.5: Fish lengths (cm) and masses (g): mean, min and max per area (summed counts)

Mullet	**Lengths (cm)**			**Mass (g)**		
	Mean	Min	Max	Mean	Min	Max
North	40.47	23	50	619.88	155	1300
Middle	37.99	1	56	833.17	0.1	1810
South	15.29	5	45	166.79	0.8	1000
Steenbras	**Lengths (cm)**			**Mass (g)**		
	Mean	Min	Max	Mean	Min	Max
North	44.25	35	52	981.41	300	2155
Middle	55.18	36	68	2948.83	795	4910
South	32.56	7	60	906.14	5.3	4060

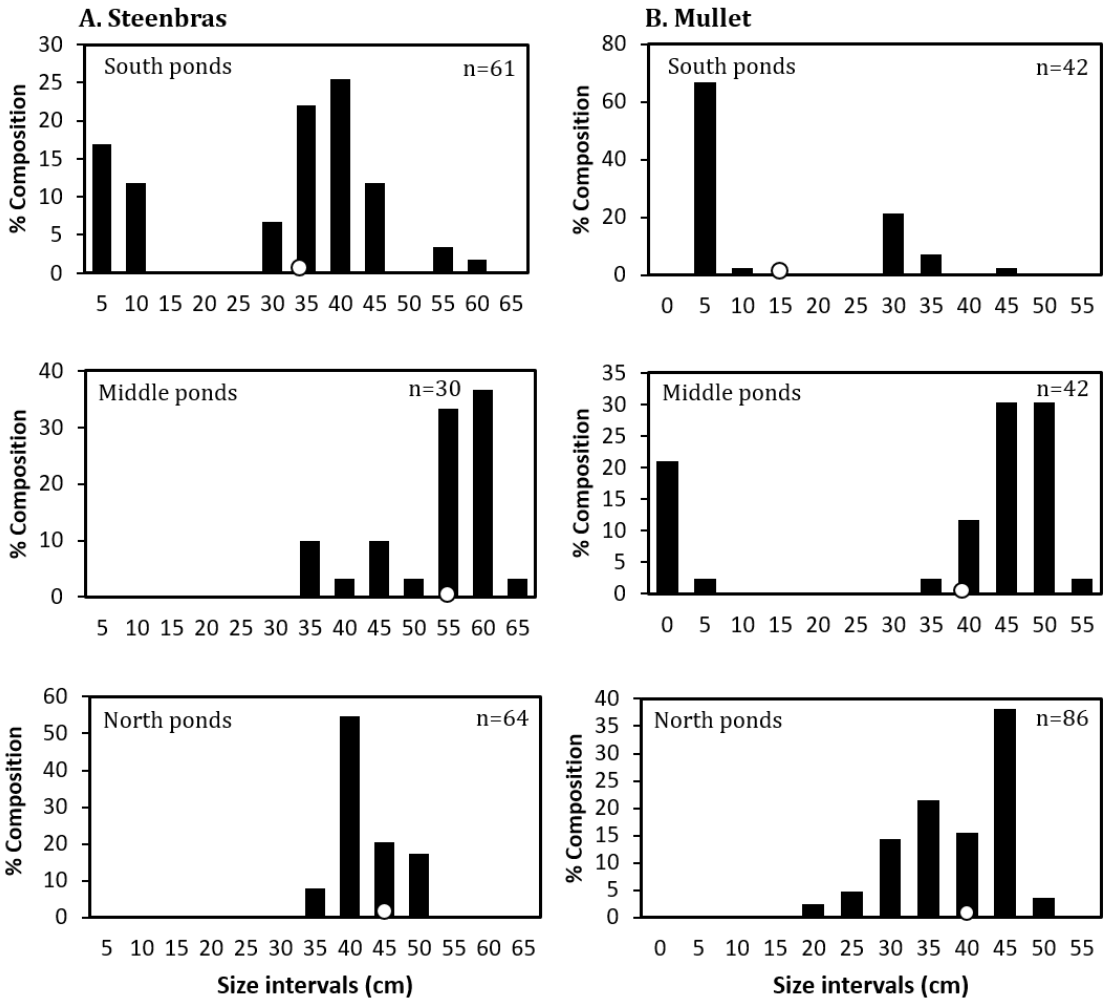

Figure 4.4: Percentage size compositions of a) Steenbras and b) Mullet. Circles indicate mean values; n = sample size. Size intervals: 0 = 0-4.9; 5 = 5-9.9 etc.

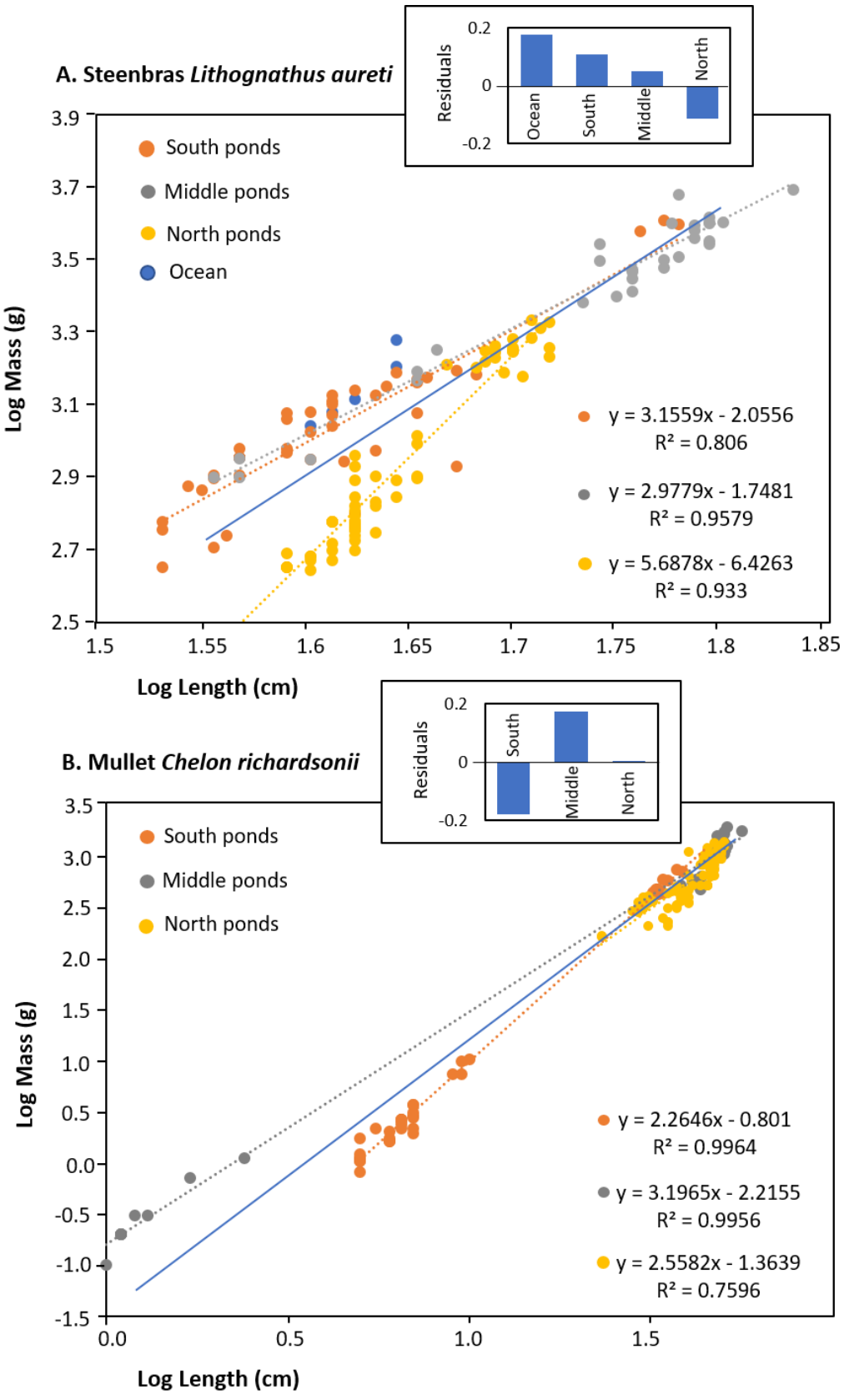

Figure 4.5: Log-log regression of body mass vs, length for a) Steenbras and b) Mullet. Solid blue lines are regressions for the whole data set. Mean residuals are inset top right

Stomach contents and gut fullness

The stomach contents of all fish caught from the mining ponds, except for juveniles that were too small to dissect, were analysed. The stumpnose guts were all empty with a maximum stomach weight of 7.1 g and minimum of 2 g. The kob stomachs were filled with diatoms (unidentified algae) and one crab (full stomach weight 13.5 g, empty stomach weight 10 g).

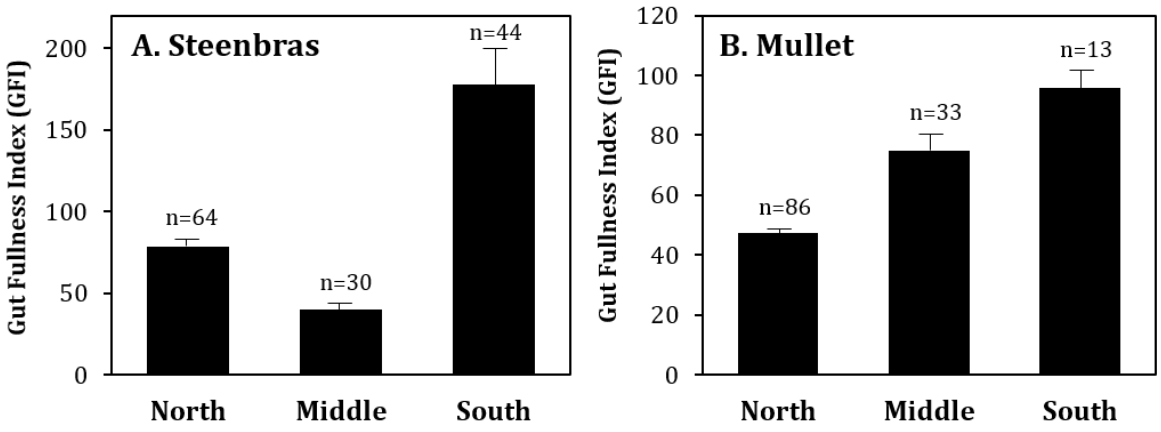

Figure 4.6: Gut Fullness Indices for a) Steenbras, b) Mullet in north, middle and south ponds; n = sample sizes

The stomach contents of 138 steenbras, excluding the 17 juveniles from the south ponds and the six caught from the ocean, were examined and the GFIs in the three pond areas determined (Fig. 4.6a). Means and ranges of the weights of the full and empty stomachs are recorded in Table 4.6. Differences in GFIs among areas were significant (ANOVA, $F_{2,135} = 6.35$, $p = 0.0023$), with values being lowest in the middle ponds and highest in the south. Post-hoc Tukey tests showed south ponds differed from both north and middle ponds ($p < 0.05$), north and middle ponds did not differ significantly from each other ($p > 0,05$).

The stomach contents of 132 mullet, excluding the 38 juveniles, were examined (Fig. 4.6b). Differences in Gut Fullness Indices (GFIs) among areas were significant (ANOVA, $F_{2,129} = 5.93$, $p = 0.0034$). GFIs in middle and south ponds did not differ (post-hoc Tukey tests, $p > 0.05$) but north ponds had significantly smaller GFIs ($p < 0.05$). The means and ranges of values for the full and empty stomachs are recorded in Table 4.6. Detritus and diatoms mixed with sediment particles dominated the guts of all mullet.

The stomachs of north steenbras contained diatoms (unidentified and unquantified) but were universally and predominantly filled with the larvae/pupae of chironomid flies. The middle ponds' steenbras stomach contents were dominated by mussel alone in 14 fish, or by mussel plus one or two unidentified fish in 16 individuals. Of the 44 steenbras sampled in the south, all had stomach contents dominated by mussel, 11 exclusively so, but 11 additionally contained crabs and 22 a single unidentified fish (< 2 cm) (Fig. 4.7). Without exception, the mullet stomachs contained only a mush of detrital particles and diatoms, which could not be separated quantitatively, and occurred universally in all individuals examined.

Table 4.6: Masses of full and empty stomachs for (a) Mullet in North (n = 86), Middle (n = 33), South (n = 13) areas, and (b) Steenbras in North (n = 64), Middle (n = 30), South (n = 44) areas

A. Mullet	**Full stomach (g)**			**Empty stomach (g)**		
	Mean	Min	Max	Mean	Min	Max
North	62.63	14.5	168	43.83	9	93.5
Middle	90.98	25	194	57.3	20	110
South	45.68	4.5	130	24.05	2.5	72
B. Steenbras	**Full stomach (g)**			**Empty stomach (g)**		
	Mean	Min	Max	Mean	Min	Max
North	49.31	19	110	27.38	11.7	50
Middle	88.45	25	165.3	63.91	1.1	136
South	48.31	2	125	23.65	0.1	90

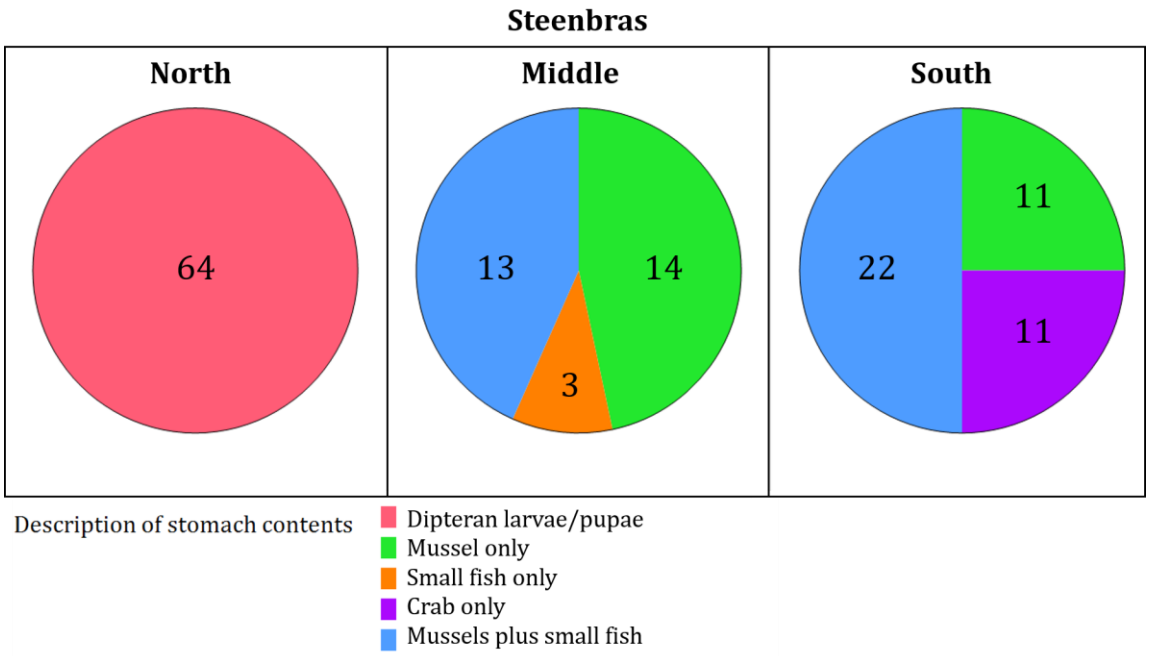

Figure 4.7: Proportions of steenbras with different prey items in their stomach contents, in north (n = 64), middle (n = 30) and south (n = 44) ponds

Gonads

Gonads of 170 mullet were examined. Of these, 38 were juveniles, and 132 were adult. For those that could be sexed, 35 were male and 97 female. The mean mullet gonad weights amounted to 16.1 g, 46.2 g and 2.6 g in the north, middle and south respectively. The mean gonad weights for males were respectively 8.0 g, 13.7 g and 0.5 g and for females 20 g, 53.4 g and 2.7 g (Table 4.7). Females significantly outnumbered males (2.9:1, chi-squared = 16.9, df = 1, $p < 0.0001$).

For mullet, stages of reproduction in the north for 41 individuals (male: 21 and female: 20) were: developing, 7 (male:1 and female:6); immature, 26 (male:6 and female:20); mature and ripe 12 (all female). In the middle ponds 1 male was developing, 1 male immature, 17 were mature (male:4 and female:13) and 14 (all female) were ripe. In the south, 6 females were developing, 6 were immature (male:1 and female:5), and 1 female was mature (Fig. 4.8a). The overall proportion of females (73.5%) was higher than the proportion of males (26.5%), so the M:F sex ratio was skewed 1:2.9 in favour of females.

Gonads of 138 steenbras, excluding 17 juveniles, were examined, of which, 21 were male, 101 female and 16 protandrous. All individuals were female in the north pond, whereas in the south pond three were female, 10 male and five in a protandrous state. Overall,

females significantly outnumbered males by 2.3:1 (chi-squared = 16.9, df = 1, $p < 0.0001$). The mean steenbras gonad masses amounted to 5.1, 24.6 and 18.7 g in the north, middle and south ponds respectively. The mean gonad weights for males were respectively 0.5, 9.1 and 14.5 g, and for females 5.2, 30.8 and 25.2 g; and for protandrous individuals, which we absent in the north, 17.5 g in middle and 12.3 g in south ponds (Table 4.8).

In the north 25 females were developing, 31 (male:1 and female:30) were immature and the remaining eight (all females) were mature. In the middle ponds, two (male:1 and female:1) were developing, one male was immature, 16 (male:3 and female:13) were mature, five females were ripe and the remaining six steenbras were in a protandrous stage of development. In the south, 13 (male:8 and female:5) were developing, seven (male:3 and female:4) were immature, one male and five females were mature, eight (male:3 and female:5) were ripe and the remaining 10 were in a protandrous stage of development (Fig. 4.8b).

Table 4.7: Mean, minimum and maximum gonad weights (g) of mullet (n = 132) for North, Middle and South

Area	All mullet			Male			Female		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
North	16.1	0.1	79.3	8	0.1	22	20	0.3	79.3
Middle	46.2	3.5	128.6	13.7	3.5	27.2	53.4	13.7	128.6
South	2.6	0.5	12.1	0.5	0.5	0.5	2.7	0.5	12.1

Table 4.8: Mean, minimum and maximum gonad weights (g) of steenbras (n = 138) for North, Middle and South

Area	All steenbras			Male			Female			Protandrous		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
North	5.1	0.3	22.1	0.5	0.5	0.5	5.2	0.3	22.1	–	–	–
Middle	24.6	0.6	90.5	9.1	0.6	19.2	30.8	5.5	53.1	17.5	0.8	90.5
South	18.7	0.3	96.2	14.5	0.6	58.1	25.2	0.3	96.2	12.3	0.7	31.9

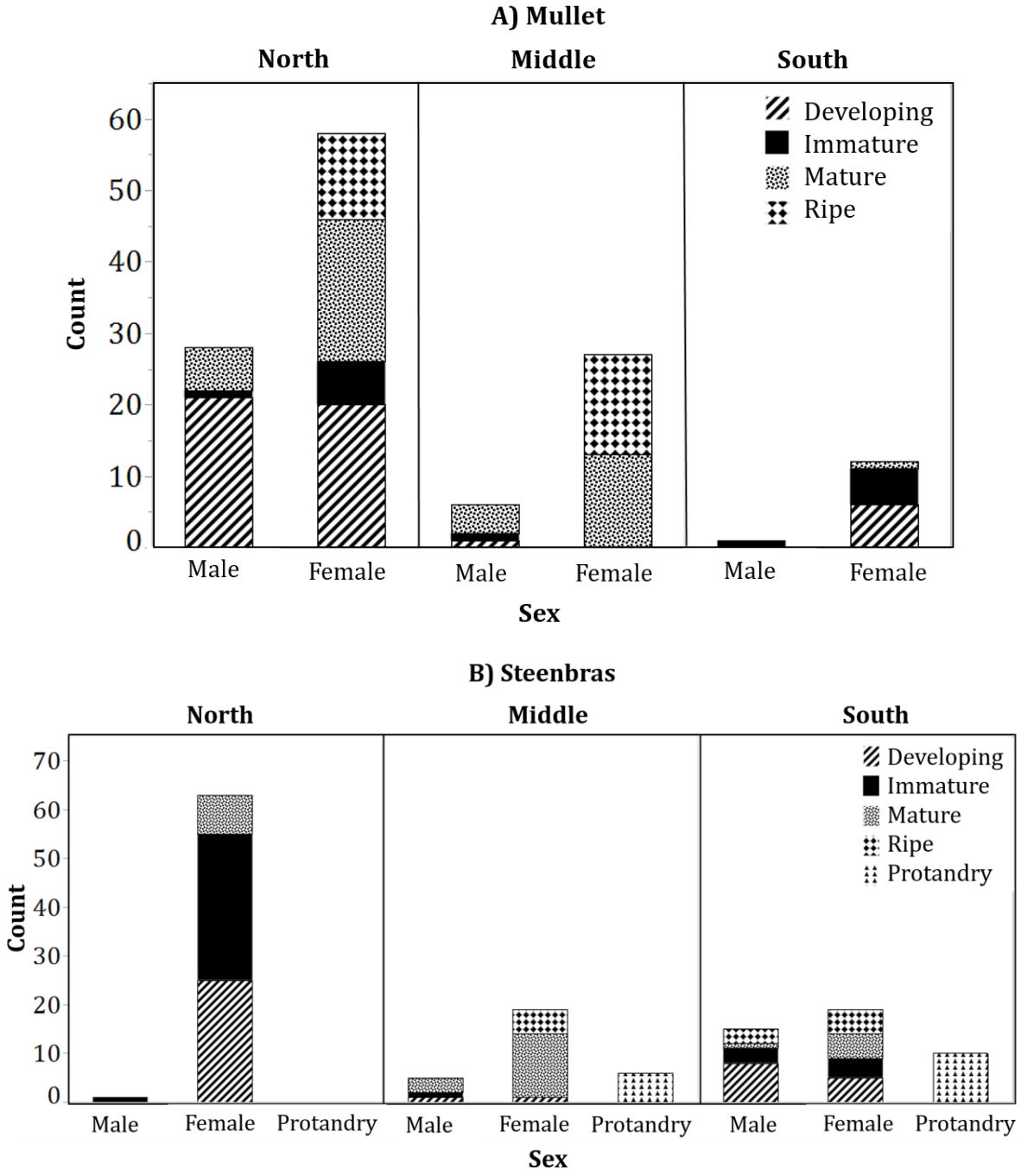

Figure 4.8: Stages of gonad development among males and females for a) Mullet (n = 132) and b) Steenbras (n = 138) in each of the north, middle and south areas

An additional three steenbras from two ponds each, S3 and N49, were caught monthly for six months between April and September 2018 and were added to the more comprehensive data for December 2017 and January 2018 to compare the gonad stages and cycles of west coast steenbras *Lithognathus aureti* between the north ponds versus

the combination of middle and south ponds (which were combined because these two areas were not sampled in all of the sampling months).

Over the course of the year, steenbras in the north ponds included individuals in immature, developing and mature stages only. In the south and middle ponds, however, all stages of development were recorded, including ripe and protandrous fish, and all individuals in June were ripe. As sample sizes were small in most months, proportions of different stages in different months should be interpreted with caution.

In January, 64% of steenbras in the north were immature whereas in the south and middle, 82% were immature in May. The highest stage of developing gonads recorded in steenbras was 28% in January in the north and 66% in April in the south and middle. In the north, 25% of steenbras were mature in December and 100% in September, whereas in the south and middle 60% were mature in December. No steenbras recorded in the north ever reached being ripe, but in the south and middle the incidence reached 100% in June and 16% in December. Only the south and middle ponds had any protandrous steenbras, with values reaching 66% in July and September, and 43% in January (Fig. 4.9). No spent/recovering steenbras were ever recorded.

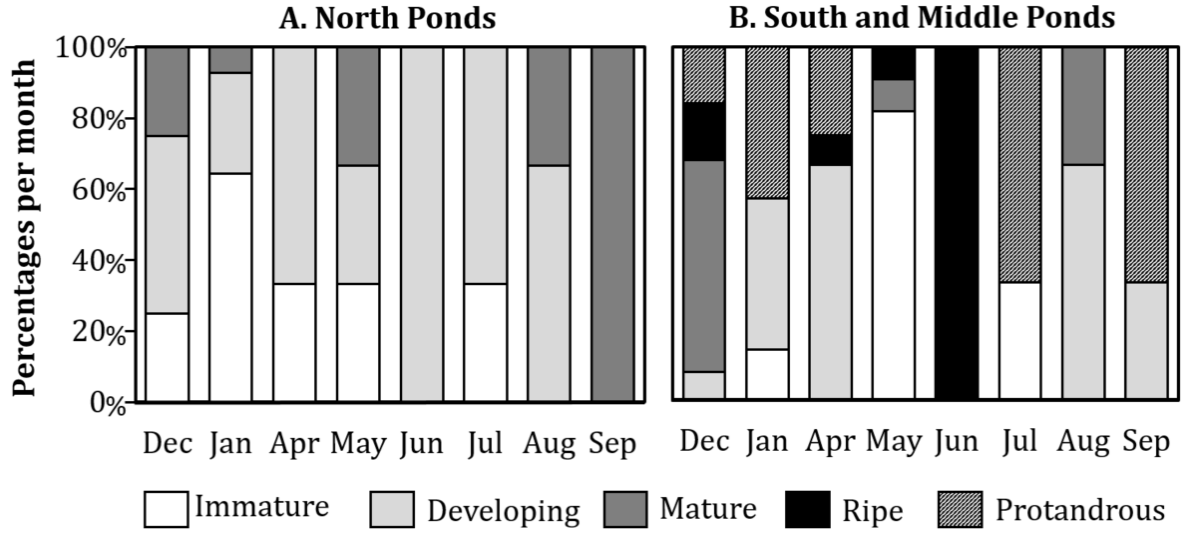

Figure 4.9: Percentages of different stages of gonadal development for steenbras over a year, for a) North (n = 64) and b) South and Middle Ponds (n = 67)

For steenbras (Fig. 4.10a), there was a clear progression from immature fish with no gonads in size classes 10-15 cm, through males, which spanned sizes 10-45 cm, protandrous individuals of 40-50 cm, to females of 35 to >65 cm. The sizes of male, protandrous and female individuals were significantly different (ANOVA, $F_{2,158} = 41.14$, $p < 0.0001$). For mullet (Fig. 4.10b), all individuals < 15 cm were immature, while males and females spanned similar ranges of 25 to 60 cm and 25 to >75 cm respectively. Despite the apparent similarity of range in males and females, the average size of females (43.9 cm) was significantly larger than that of males (37.6 cm) ($t\text{-test}_{131} = 3.75$, $p = 0.0002$).

The gonadosomatic indices (GSIs) for steenbras were consistently low in north ponds, but escalated to a peak in June in the south-plus-middle ponds (Fig. 4.11). This coincided with the appearance of ripe gonads (Fig. 4.9). The effects of both month and area on the GSI were significant (two-way ANOVA on balanced design, $F_{7,32} = 15.95$, $p < 0.0001$ and $F_{1,32} = 25.14$, $p < 0.0001$ respectively), but there was a significant interaction ($F_{7,32} = 17.17$, $p < 0.0001$) arising from the ascendency of the GSI in June in the middle ponds.

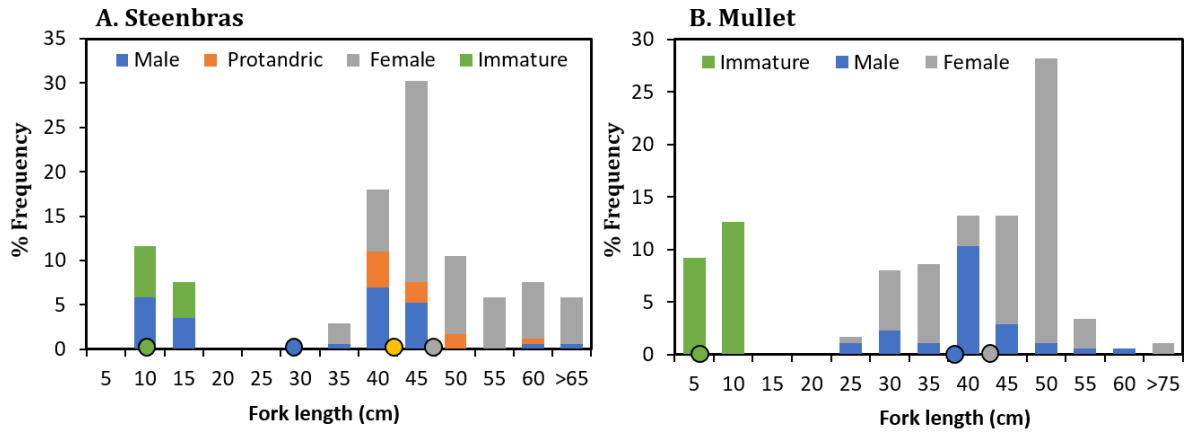

Figure 4.10: Percentage size frequencies of a) Steenbras (n=161) and b) Mullet (n=170), in relation to state of gender. Circles indicate mean values. Size class 5 = 5-9.9 cm, 10 = 10-14.9 etc.

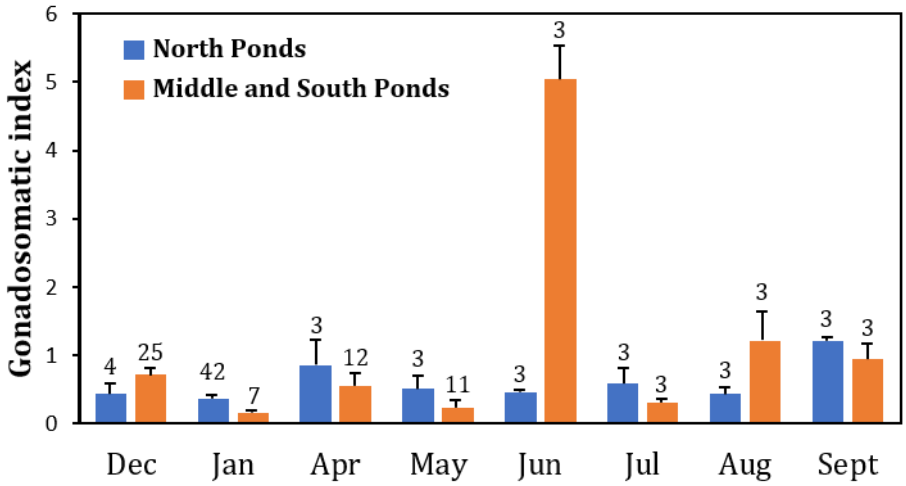

Figure 4.11: Gonadosomatic indices (means +SE) for steenbras in North ponds (n=64) and in Middle (M) and South (S) ponds (n=67) combined. Numbers above bars indicate sample sizes

Growth

Otoliths were collected from only southern mullet and west coast steenbras, and used to examine growth rates. From the 70 otoliths collected, 60 were readable. The mullet sample consisted of 19 otoliths: three from south ponds and 16 from North ponds. Mullet from pond N49 were between 3 and 5 years old, and pond N116 had the oldest mullet with ages ranging from 1 to 8 years (Fig. 4.12a).

The steenbras sample comprised 41 otoliths: 18 from south ponds, nine from middle ponds, nine from the north, and five from the ocean. Steenbras caught from the south ponds ranged between 1 and 11 years although those caught from S17 were all under a year. Middle ponds had the oldest steenbras ranging between 10 and 30 y (Fig. 4.12b). The steenbras caught from the ocean ranged between 1 and 8 y.

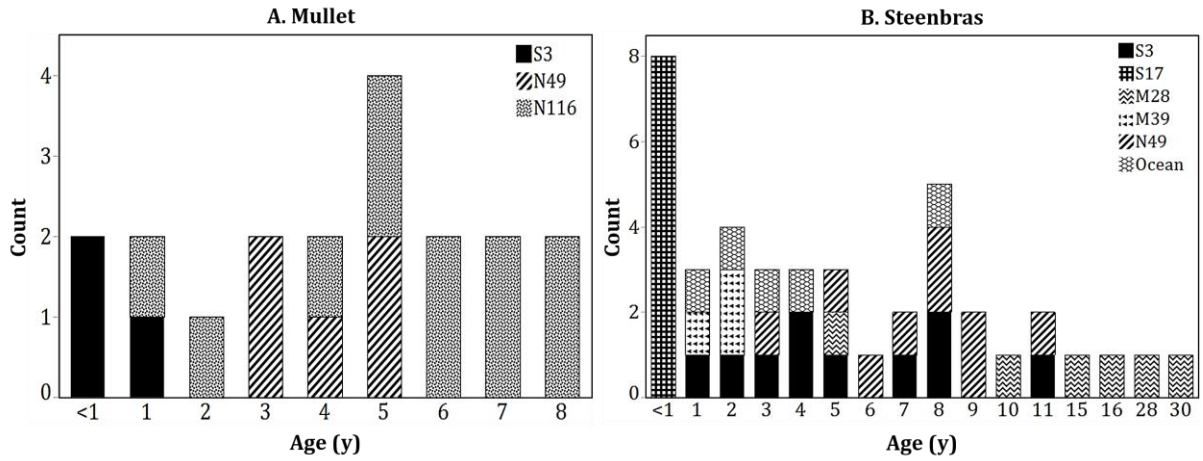

Figure 4.12: Age of a) Mullet (n = 19) and b) Steenbras (n = 41), for individuals whose age was determined from otolith examination

Ford-Walford plots of L_{t+1} versus L_t were derived for both steenbras ($L_{t+1} = 0.4819L_t + 29.03$; $R^2 = 0.58$) and mullet ($L_{t+1} = 0.386L_t + 29.69$; $R^2 = 0.97$). From these, Van Bertalanffy equations and curves were calculated for both species (Fig. 4.13), based on pooling all available samples. There were too few data to statistically compare among areas, but samples of steenbras from north, south plus middle and the ocean clustered together with no indication of obvious differences in growth among areas. For steenbras, the Ford-Walford plot ($y = 0.4819x + 29.03$, $R^2 = 0.5753$, $p < 0.05$) yielded $K = 0.730$, $L_\infty = 57$ cm, with maximum sizes actually recorded being in close agreement at 60-62 cm. For mullet, the Ford-Walford plot ($y = 0.386x + 29.69$, $R^2 = 0.97$, $p < 0.0001$) predicted $K = 0.951$, $L_\infty = 48$ cm, coinciding almost exactly with the recorded largest value of 49 cm.

Using the growth curves, the data for size compositions could be converted to estimated ages, yielding a larger data set of ages (Fig. 4.14). This confirmed that recruits (< 1 y) of mullet were confined to south and middle ponds, and those of steenbras to south ponds only. For both species maximum and mean ages were greatest in middle ponds (and this remained true even if the biasing effect of recruits was removed by excluding them from consideration). Of particular note is the fact that the north ponds housed a wide range of ages: 3-12 y for steenbras and 1-16 y for mullet.

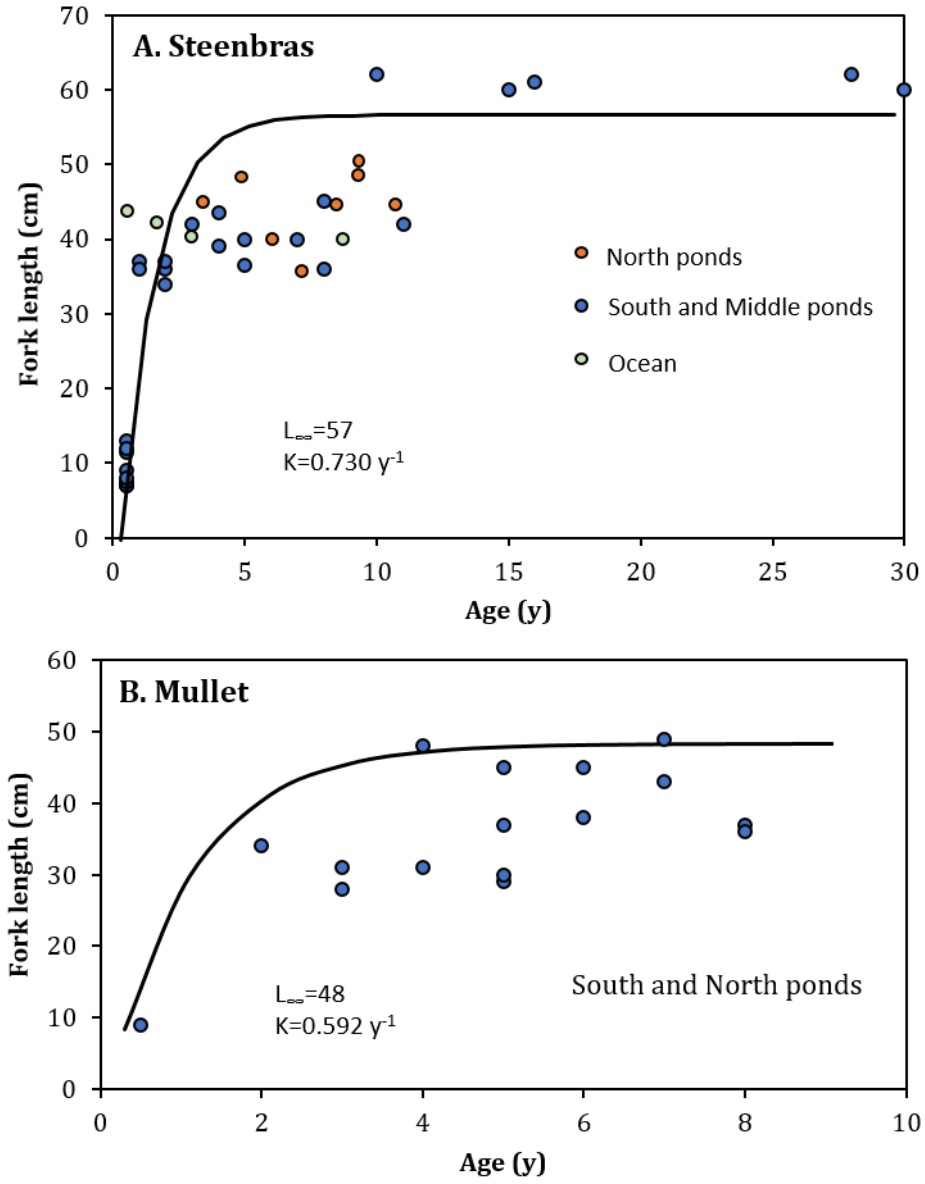

Figure 4.13: Von Bertalanffy growth curves for a) Steenbras, fitted to the data for all ponds combined but excludes the ocean data, which are superimposed for comparison, and for b) Mullet, for which the curve incorporates data from both south and north ponds

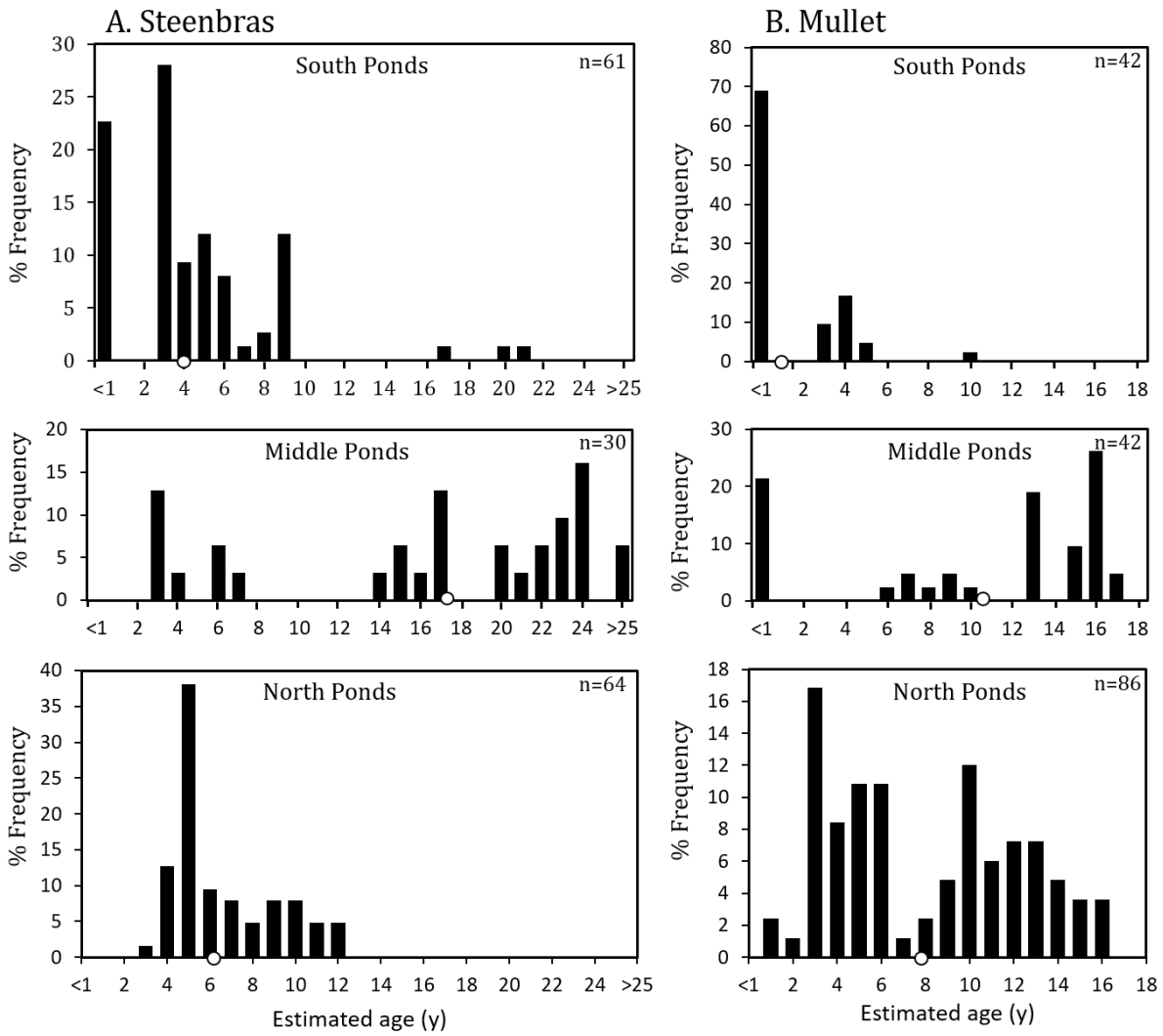

Figure 4.14: Age composition (percentage frequency) of a) Steenbras, and b) Mullet for north, middle and south ponds. Circles are means; n = sample size. The values are estimates derived from length compositions, based on the von Bertalanffy growth curves

DISCUSSION

The coast of South Africa has been divided into three biogeographical zones based on their associated estuarine fish assemblages: (1) a cool-temperate region extending from the Orange River, just before the study area, down the west coast and along the southwest coast to Cape Agulhas; (2) a warm-temperate region extending from Cape Agulhas to just south of Port St Johns and (3) a subtropical region extending up the east coast from Port St Johns (Harrison and Whitfield 2006).

Building on Whitfield's (2005) assessment of estuarine fish biogeography, Potts et al. (2015) argued that these biogeographical zonation patterns are too wide and concentrated on South Africa, and further reviewed the regions along the south-west southern African coastline. They identified seven coastal biogeographical regions in southern Africa, which include (1) tropical and subtropical Angolan waters, (2) a warm-temperate west coast in northern Namibia, (3) a northern cool-temperate west coast extending to Lüderitz, (4) a southern cool-temperate west coast, (5) a warm-temperate south coast, (6) the subtropical east coast and (7) a tropical east coast. With these classifications, the Namibian coastline hosts two main biogeographic zones, a cool-temperate region northwards from Lüderitz, and a second cool-temperate region southwards from Lüderitz (Emanuel et al. 1992; Potts et al. 2015).

My study area falls within the cool-temperate zone along the south-western Namibian coastline, within the Benguela system, where there is a paucity of estuaries (Potts et al. 2015), although the Orange River flows into the Atlantic Ocean immediately to the south of the study site (Sakko 1998; Currie et al. 2008). Relative to warm temperate and subtropical systems in the region, fish species richness and abundance are low in both the Orange River estuary and the cool-temperate zones in general (Harrison 2003; Harrison and Whitfield 2006). Nevertheless, Seaman and van As (1998) found that the mouth of the Orange River Estuary had a considerable variety of fish species relative to other regional estuaries, all of which were marine species due to the salinity levels there being similar to those in the sea for up to 2 km from the mouth during high tide. In addition, the nutrient-rich Benguela Ecosystem supports important fish stocks that form the basis for the Namibian fisheries industry (Bethune and Roberts 1991; O'Toole 1997; Harrison and Whitfield 2006; Hutchings et al. 2009).

Estuaries are important nursery systems for fish, with the Orange River Estuary supporting both freshwater species and freshwater-tolerant estuarine-dependent marine species of 36 fish species representing 19 families, with eight being marine species (Lamberth 2013). These include the four main species found in the mining ponds: southern mullet *Chelon richardsonii*, west coast steenbras *Lithognathus aureti*, white stumpnose *Rhabdosargus globiceps* and silver kob *Argyrosomus inodorus*.

Given this background, the first striking result from my studies was that the ponds are depauperate in fish species. Only four species were recorded in the dedicated sampling of ponds, and around eight species in total if incidentally-recorded species are included. In the middle and north ponds, only two species were captured during dedicated sampling. Species richness and diversity were thus very low. Other surveys of estuarine systems in the region by Harrison (1998) have also revealed depauperate fish faunas. In Verloren Vlei near Lamberts Bay, only four species were recorded, with two species of mullet, *Chelon richardsonii* and *Mugil cephalus*, being dominant. In the Berg River Estuary 12 species were documented, with *C. richardsonii* again being dominant. Olifants and the Orange River estuaries hosted nine species each. Although Lamberth's (2013) more detailed survey of the Orange River detected 36 species, only eight were marine. Estuaries in the region are thus species-poor in marine fish, and the diamond-mining ponds are either on a par with these systems, or even more depauperate.

Guilds

Elliott et al. (2007) developed a classification of guilds for estuarine usage functional groups (EUFGs) for fish in estuarine environments, by describing and interpreting the community structure and function of fishes inhabiting estuaries. Like natural closed estuaries, the mining ponds will undergo changes of temperature, dissolved oxygen, salinity, pH and chlorophyll depending on evaporation rates, rainfall (which is minimal) and, most importantly, seawater inputs when seawater overtops the seawall, enters through seepage, or when there are periodic occurrences of breaching.

Using Whitfield's (1994) classification of fish species by their degree of dependence on estuaries during their lifecycle (Table 4.9), and the guild approach of Elliot et al. (2007) to the functional attributes of fish communities allows us to understand the ecological role of the ponds and gain insight into natural (water quality) and anthropogenic (mining) stressors.

The estuarine usage functional groups (EUFGs) comprise four categories, namely marine, estuarine, diadromous and freshwater, with each containing multiple guilds. Potter et al. (2015) further subdivided the four categories of Elliot et al. (2007) into 14 groups. According to these classifications, the four fish species inhabiting the ponds all fall under

the marine category. The three guilds in the marine category range from species that occur accidentally in estuaries (marine stragglers) to those that tend to enter estuaries in large numbers at some stage in their life cycle, typically during their juvenile stage (marine estuarine-opportunists) and those that depend on these systems for survival at a critical stage in their life cycle (marine estuarine-dependent species) (Elliott et al. 2007; Potter et al. 2015).

Marine stragglers and estuarine-opportunists are regularly found along the southern African coast, where the highly exposed waters are unfavourable for fish during their juvenile phase, which relies on the protected waters of estuaries to provide a suitable nursery habitat (Blaber 1981; Whitfield 1994). Similarly, the fish inhabiting the ponds make use of these sheltered environments, and may spend their entire lives in the ponds unless a breach occurs, allowing fish back into the sea.

Table 4.9: The five major categories and subcategories of fish that utilize southern African estuaries by the degree of dependence on estuaries during their lifecycle (Whitfield 1994)

Category	Description
I	Estuarine species that breed in southern African estuaries. Ia – Resident species that have not been recorded spawning in marine or freshwater environments. Ib – Resident species that also have marine or freshwater breeding populations.
II	Euryhaline marine species that usually breed at sea, with juveniles showing varying degrees of dependence on southern African estuaries. IIa – Juveniles dependent on estuaries as nursery areas. IIb – Juveniles occur mainly in estuaries, but are also found at sea. IIc – Juveniles occur in estuaries, but are usually more abundant at sea.
III	Marine species that occur in estuaries in small numbers, but are not dependent on these systems.
IV	Freshwater species, whose penetration into estuaries is determined primarily by salinity tolerance. This category includes some species that may breed in both freshwater and estuarine systems.
V	Catadromous species that use estuaries as transit routes between the marine and freshwater environments, but may also occupy estuaries in certain regions. Va – Obligate catadromous species that require a freshwater phase in their development. Vb – Facultative catadromous species that do not require a freshwater phase in their development.

As noted above, the ponds are colonised predominantly by west coast steenbras *Lithognathus aureti* and southern mullet *Chelon richardsonii*, and to a lesser extent white stumpnose *Rhabdosargus globiceps* and silver kob *Argyrosomus inodorus* (Fig. 4.15), and accounts of these species follow below.

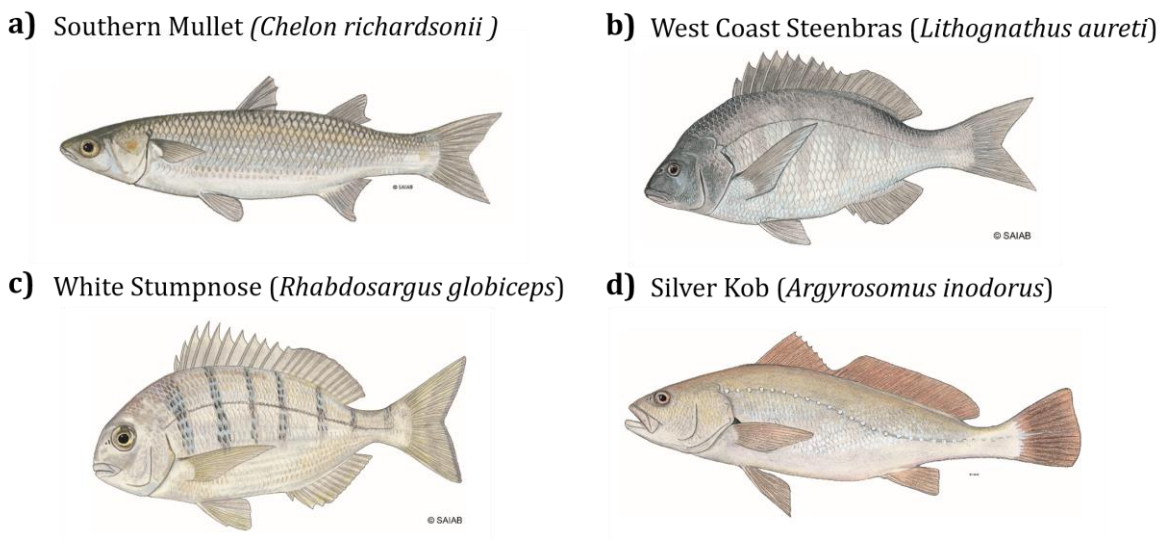

Figure 4.15: Main fish species inhabiting the mining ponds. Images from WWF Southern African Sustainable Seafood Initiative (SASSI) courtesy of SAIAB: http://wwfsassi.co.za

Southern Mullet (*Chelon richardsonii*)

The southern mullet is endemic to the southern African coast, spawning close inshore (Lasiak 1983), with juveniles frequently occupying estuaries but also common at sea (Beckley 1985; Romer and McLachlan 1986; Clark et al. 1994; Whitfield 1994 2019) and may be even more abundant in the sea (De Villiers 1987), thus rendering it a Category IIb or IIc species (Table 4.9). *Chelon richardsonii* is a nomadic coastal, bottom-dwelling fish found in the surf zone, nearshore marine environments and lagoons and estuaries from Lobito (Angola) through the cool-temperate west-coast zones to the warm-temperate east coast (Lasiak 1983; DAFF 2014). Cambray and Bok (1989) have described the early development of the species, which is commercially exploited and 25-40% overexploited (DAFF 2014). Its conservation status has not been evaluated (IUCN 2019).

De Villiers (1987) conducted a study on the southern mullet in the Benguela upwelling region. He described the species as the most abundant of the 15 mugilid species reported from South African waters and one of only two species that can be found from southern KwaZulu-Natal to northern Namibia and Angola. Although there is limited information available on the growth rate of the southern mullet, De Villiers (1987), Lasiak (1983) and Horton et al. (2019) agree that it matures at 18-22 cm, which concurs with my findings: only individuals > 21 cm were mature. In the marine embayments of Saldanha Bay and Langebaan Lagoon, Horton et al. (2019) found significantly differences in growth among

the sexes, with females growing slower but reaching larger sizes. My data also showed that females reached larger sizes. Pooling the Saldanha/Langebaan data for the sexes, estimates of L_∞ = 27.1 cm and K = 0.51 y^{-1} were obtained. Earlier values for the marine population in False Bay were L_∞ = 35.9 cm, K = 0.256 y^{-1} (Ratte 1977), and L_∞ = 35.2 cm, K = 0.287 y^{-1} (De Villiers 1987). My data for *C. richardsonii* in the ponds are based on a small sample size, but they consistently predict double the growth rate and almost double the maximum size compared to the Saldanha/Langebaan population, with False Bay individuals being intermediate. No data are available for growth rates of *C. richardsonii* in open-sea populations in the vicinity of my study.

Lasiak's (1983) observation that surf-zone populations in the Eastern Cape are male-dominated, with a M:F ratio of 2.1:1 is the opposite to what I found, as mullet in the ponds had a female-dominated ratio of 1:2.9. The sizes of mullet recorded in the ponds were unusually large: 42% exceeded 40 cm, with the largest being 51 cm, considerably above the normal maximum of ca. 32 cm (Whitfield 2019). Bickerton (1981) has previously noted that *C. richardsonii* living in seawater-filled trenches excavated for diamond mining at Holgat on the west coast of South Africa also grew to exceptional sizes. As females are on average larger than males (Lasiak 1983), the larger sizes of mullet in the ponds may be associated with the switch in sex ratios that I observed. The large sizes achieved may, however, simply reflect good foraging conditions, or could also be associated with the fact that predatory fish (this chapter) and piscivorous birds capable of eating large fish (chapter 3) were rare or absent. Whatever the case, the high growth rates and large sizes attained in the north ponds are notable, as they take place in the face of relatively low chlorophyll-*a* levels (chapter 1).

De Decker and Bennett (1985) showed that individuals of *C. richardsonii* in the Bot River Estuary have a better body condition (higher lipid and energy content) than those in the open-ocean waters of False Bay. The body condition of the mullet I sampled varied from good in the middle ponds, neutral in the north, and poor in the south.

Seaman and van As (1998) and Lamberth (2013) described the southern mullet *Chelon richardsonii* as the dominant species within the Orange River estuary. It was the species that was dominant in the north ponds (51% of all mullet were recorded there) but was

also abundant in the south and middle ponds. It feeds mostly on benthic detritus and diatoms or suspended particles (Masson and Marais 1975; De Villiers 1987; Lamberth 2013; Whitfield 2016) and all mullets caught from the ponds had stomach contents that were filled with detritus and diatoms mixed with sand, which assists with crushing food in the stomach (Masson and Marais 1975). The north ponds' salinities were above 70‰, with the highest salinity levels being recorded in ponds that had only mullet present (N102:78.19‰, N109:88.97‰ and N116:85.05‰) (Chapter 1, Table 1.1).

The southern mullet is an opportunistic species tolerant of both hypo- and hyper-salinity, and has been recorded in salinities ranging from 2-90‰ (Whitfield 2016 2019), and is therefore able to survive in the saline northern ponds that have been isolated from the sea for an extended period. Indeed, there was no significant relationship between mullet numbers per pond and the salinities of ponds, reflecting their wide tolerance.

There is evidence that the gonads of the southern mullet were at different developmental stages in the ponds, including immature, developing, mature and ripe, suggesting that the species can breed in the mining ponds. Juveniles < 1 year old were found in south ponds (particularly S3, the mariculture pond). Mullet between 3 and 8 years old (or 1 to 8 y, if one accepts the age estimates based on size composition) were also found living in the north area, where mining activities ended more than 15 years ago and the ponds have been isolated from the sea, further supporting the idea that they breed in these ponds. It must, however, be conceded that no 'spent' individuals were recorded in the ponds, so it is possible that ripe gonads were simply resorbed without any spawning taking place. This would, however, leave unexplained the presence of recruits and juveniles in ponds that have been isolated from the sea for prolonged periods. Post-larvae of several fish, including mugilids and Cape stumpnose *Rhabdosargus holubi* have been recorded entering the Kleinemonde Estuary on the south coast of South Africa during its closed phase when water overtops the berm (Cowley et al. 2001, Kemp and Froneman 2004). This is, however, an unlikely mode of entry to the north ponds, which were not created by accretion (see chapter 1) and lie far from the sea, so that breaching and overtopping have never been recorded there, making recruitment from the sea highly unlikely. This stands in contrast to the southern and middle ponds where these events are more likely.

West Coast Steenbras (*Lithognathus aureti*)

The West coast steenbras is largely confined to seawater, entering estuaries only as a 'marine straggler', and is not dependent on estuaries (falling into Category III in Table 4.9) (Whitfield 1990, 1994). It is endemic to the southern African coast from southern Angola to Cape Town but is abundant only in Angola and Namibia. Two separate stocks are found in Namibian waters, namely a closed population in the southern region adjacent to where I worked, and a separate population in the central and northern regions (Holtzhausen and Kirchner 2001). It is an important recreational fishery species in Namibia (Holtzhausen et al. 2001) and is listed as 'near threatened' on IUCN's list of threatened species (IUCN 2019).

Holtzhausen and Kirchner (2001) studied the age and growth of the two populations of west coast steenbras in Namibian waters and consider it especially vulnerable to overfishing due to its slow and prolonged growth rate, with the northern population reaching ages of more than 50 years and the southern population reaching ages of at least 32 years. They suggest that the geographical differences in growth rates may be a result of differences in sea temperature and food availability. The von Bertalanffy curves these authors provide for growth in the southern and northern populations of *L. aureti* indicate that sizes of respectively around 25-37 cm and 32-52 cm are achieved after 5 years, and that asymptotic lengths (L_∞) there are respectively 73 and 84 cm (Beyer et al. 1999; Holtzhausen and Kirchner 2001). My data for ponds are admittedly from a much smaller sample size, but they indicate a much faster growth rate than the southern population, of around 56 cm after 5 y (predicted from the van Bertalanffy growth equation) and a span of data points covering 36-48 cm for this age. That would place the pond growth in the range of the northern population. The predicted L_∞ of 57 cm I calculated for the pond population is, however, much less than either of the values recorded by Holtzhausen and Kirchner (2001) for the two ocean populations. It is of interest to note that Wallace (1974) recorded *Pomadasys commersonnii* (a similar zoobenthivore to *L. aureti*) feeding mainly on filamentous algae in the northern parts of Lake St Lucia where hypersaline conditions prevailed and zoobenthos was scarce. It is possible that the filamentous algae were ingested during the targeting of chironomid larvae associated with the St Lucia algal mats.

Lucks (1970; cited by Mehl 1973) recorded growth rates for *L. aureti* in Sandwich Harbour, Namibia (which experiences salinity levels close to those of seawater: 34.9 to 35.4‰) that equate to around 17 cm.y^{-1} for the first three years of life, and then 7 cm.y^{-1} thereafter. Although Mehl (1973) challenges the validity of these data, they agree closely with the rates I determined from otolith samples. The maximum size of *L. aureti* in Sandwich Harbour was 77 cm (Lucks 1970), with peak frequencies at 32-36 cm and 58-66 cm, which would follow the patterns I detected if I pooled the data for all ponds and excluded juveniles < 20 cm. Sizes at Meob Bay, a protected area where no fishing takes place, spanned 20-70 cm (Beyer et al. 1999).

During tag-and-release expeditions by Holtzhausen and Kirchner (2001), biological data were collected, including length, sex and sagittal otoliths, which were used to determine growth patterns. Stomach contents of fish in the southern population indicated that black mussels (*Choromytilus meridionalis*) and brown mussels (*Perna perna*) were their preferred food. However, during their study, more than half of the southern population's stomachs were empty which suggests food availability is less in the south and may be responsible for the slower growth there compared to the northern population, which had only approximately one third empty stomachs during the study. My results reflect a similar diet in the south and middle ponds, comprising mussels, small fish and crabs, but in the north ponds the diet was radically different and more uniform, comprising chironomid fly larva and pupae, as well as diatoms that may have been ingested incidentally while consuming these benthic insects. Lucks (1970) recorded that in Sandwich Harbour *L. aureti* consumed amphipods, polychaetes and '*Mytilus*' – though in reality that is likely to have been another species of mussel, most probably *Perna perna*, as this record predates the recorded arrival of *Mytilus galloprovincialis* in southern Africa in about 1979 (Grant and Cherry 1985).

Lithognathus aureti occurred predominantly in the north ponds (41.3% of 155 individuals sampled) where mining activities ended more than 15 years ago and salinities are now high. The south ponds also had high steenbras numbers (39.3%) and the middle ponds the least numbers (19.4%). The steenbras in the north ponds were in poor condition compared to those in the south and middle ponds, and condition was best in

the ocean; but as the sample size was small for the ocean, this outcome should be treated with caution.

The north ponds give an indication of what can be expected for steenbras trapped inside ponds once mining has ended and the ponds age, with concomitant increases in salinity. As outlined in Chapter 1, salinity levels in the ponds remain tolerable for about 10 years but after that, they will climb steeply to intolerable levels, above 70‰, causing osmoregulatory stresses in marine organisms, and particularly fish (Whitfield et al. 2006) trapped inside the ponds. While steenbras have survived in the high-salinity north ponds for a prolonged periods, as reflected by their ages there, their numbers were exponentially negatively related to salinity.

Whitfield (2019) has synthesised information on biotic responses to salinity in St Lucia Estuary in South Africa and discusses how many fish species entering southern African estuaries have the ability to adapt to both low and high salinity regimes, but that very few fish species are able to tolerate salinity values between 70 and 110‰ (Whitfield et al. 2006). Therefore, the most essential adaptation by fish that enter the mining ponds will be their ability to adjust to changes in salinity as it increases with the age of ponds. For ponds less than 10 years old, salinity levels remained close to those of seawater; but thereafter (notably in the north ponds) values rose steeply to as much as 116‰ (Chapter 1, Table 1.1), exceeding the tolerance levels of most estuarine and marine species (Whitfield et al. 1981).

Whitfield (2019) has also summarised the combined effects of temperature and salinity in relation to fish abundance and diversity in estuaries. The extent that fish can survive before salinity levels become lethal is influenced by temperature (Whitfield et al. 1981). Although temperature was higher in all ponds than in the sea (Appendix 4.1; Table 1.1), at least some species of fish and macroinvertebrates survived in the ponds. Furthermore, in subtropical estuaries of southern Africa, temperatures do not fluctuate greatly in the short term, and even in summer usually range from 24-28°C, which suggests the temperatures recorded in the ponds will not necessarily limit the fish life there (Whitfield 2019).

No records were found of the salinity tolerance of west coast steenbras, but they were found in only two of the north ponds, N44 and N49, with salinities of 70.77‰ and 70.79‰ respectively, and were absent from the other north ponds, which had salinities of 78–89‰ (Appendix 4.1). This raises an interesting point, that as the West coast steenbras is considered a 'marine straggler' that does not regularly occupy or depend on estuaries, it would be expected to be stenohaline, with a narrow salinity tolerance. The fact that it survives as salinities as high as 70‰ places it in a euryhaline bracket and raises the question of why it does not make more extensive use of estuaries. Perhaps it cannot tolerate low salinities – a possibility that would be amenable to future experimental.

The steenbras from the north ponds were emaciated and their stomachs filled with only diatoms and chironomid fly larvae/pupae (which are able to survive at high salinity levels), in contrast to the south and middle with much lower salinities (35.3-40.6‰), where wider ranges of benthic invertebrates were recorded in both the benthos (J. Cloete, pers. comm.) and in the steenbras stomach contents. Velasquez (1992) has similarly recorded how benthic prey are larger and more diverse in saltpans with moderate salinities, but reduced in diversity and dominated by chironomids in high-salinity pans.

To survive, fish trapped in the mining ponds have to adapt to changes in salinity particularly during periods when food sources, such as macroinvertebrates that can only tolerate salinity levels up to 60–65‰ (Wooldridge et al. 2016) die off, further decreasing the viability of fish (Whitfield et al. 2006; Whitfield 2019).

As outlined in Chapter 3, inside the middle ponds, insect larvae and pupae (Chironomidae), marine isopods (*Exosphaeroma* spp.), amphipods (*Jassa falcata*) and alien invasive mussels (*Mytilus galloprovincialis*) were recorded (J. Cloete, pers. comm., and pers. obs.). The southern ponds, particularly the dewatered ponds, had crown crabs (*Hymenosoma orbiculare*), shrimp (*Palaemon peringueyi*), pipefish (*Syngnathus temminckii*), small fish (gobiid and clinid species), serpent eel (*Ophisurus serpens*), Namibian cushion star (*Asterina stellifera*), Benguela compass jellyfish (*Chrysaora fulgida*) and abundant alien invasive mussels (*Mytilus galloprovincialis*) (pers. obs. and see Fig. 3.10 in Chapter 3).

Previous work has established that west coast steenbras are protandrous (Buxton and Garratt 1990; Holtzhausen and Kirchner 2001), and my results confirm this. Thus far, it is the only southern African sparid known to be protandrous (Buxton and Garratt,1990; Lucks 1970). In the mining ponds, their gonads spanned a range of different developmental stages in various months, and the size composition data indicated that males predominated in small individuals, protandry in intermediate-sized fish, and females among large individuals. Lucks (1970) showed that individuals of *L. aureti* at all stages of reproduction occurred in Sandwich Harbour, including fully ripe specimens, and concluded that it spawned within the lagoon, which has physical conditions similar to those of the southern and middle ponds, with temperatures and salinities approximating those of seawater. There were indications in my data that the species breeds in the ponds, with juveniles of 1 year old or less being recorded in two south ponds. In the north, where mining activities ended more than 15 years ago, fish between 3-11 years old were recorded, despite these ponds being isolated from the sea for periods much greater than the age of these fish. Although spawning by marine species is unusual in estuaries, Connell (1996) has recorded incidences of spawning by other sparids (and mugilids) in St Lucia Estuary, Durban Harbour and Kosi Bay. As was the case for mullet, the absence of 'spent' individuals of steenbras in my data may, however, argue against breeding taking place in the ponds. Steenbras from one of the middle ponds (M39) were between 1 and 2 years old. A breach event occurred at M39 on September 2018 (see Fig. 1.6, Chapter 1) and flooded that mine area, creating a pond and trapping fish that entered from the sea inside. Another middle pond (M28) contained the oldest individuals, ranging from 10 to 30 years. M28 was created recently when the shoreline was pushed seawards causing fish from the sea to become trapped inside the newly created pond.

The fish from the south and middle ponds will survive until mining is initiated and entire ponds are dewatered and mined out, which will destroy all marine life inside until refilling of the ponds allows recolonisation.

White Stumpnose (*Rhabdosargus globiceps*)

According to Whitfield (1994) juvenile stumpnose are dependent on estuaries for nursery grounds and although adults can be found in estuaries (Category IIa in Table 4.9),

they generally migrate back to sea once they reach sexual maturity at 23 cm for females and 27 cm for males (Buxton and Kok 1983; Whitfield 2019).

White stumpnose is an endemic species found in the shallow marine waters along southern Africa with juveniles feeding on zooplankton, isopods and filamentous algae. Once they develop molars, adults feed on benthic macroinvertebrates (Buxton and Kok 1983; Whitfield 1998). It is listed as 'vulnerable' on IUCN's (2019) list of threatened species.

Only six stumpnose (all from S16) were caught during the study with their lengths ranging between 20 cm and 28 cm. All had empty stomachs.

Silver Kob (*Argyrosomus inodorus*)

The silver kob, a migratory species, spawns along the Namibian coast between Sandwich Harbour and Meob Bay (Holtzhausen and Kirchner 2001) and feeds mainly on pelagic fish (Griffiths 1996). This marine species occurs in estuaries in small numbers but is not dependant on these systems (Category III in Table 4.9) (Whitfield 1994).

The silver kob is endemic to southern Africa from northern Namibia to southern Transkei, and is described by Potts et al. (2015) as a slow-growing and early-maturing species. The silver kob's stock status is collapsed to about 25% and it is currently considered as overexploited and overfished in South Africa (DAFF 2014). The conservation status of the silver kob has not been evaluated (IUCN 2019).

Griffiths (1996) studied the age and growth of the South African silver kob from Cape Point to the Kei River. Throughout the study there were very few kob caught over the age of 13 years, even though their maximum age is 25 years, and Griffiths (1996) attributed the absence of old individuals to overexploitation.

Only two silver kobs, one from S23 (93 cm fork length) and the other from the ocean (71 cm fork length), were caught during the time of the study. Hypersalinity would also have a direct negative influence on fish growth rates as considerable energy would be required for osmoregulation.

CONCLUSIONS

The two dominant species in the mining ponds, west coast steenbras and southern mullet, both endemic to the southern African coast, are important commercial and recreational species in Namibia. Southern mullet spawn at sea and enter estuaries in large numbers as juveniles (O'Toole 1997; Bianchi et al. 1999; Boyer and Hampton 2001; Holtzhausen et al. 2001; Elliot et al. 2007), and are classed as 'estuarine dependent'. West coast steenbras normally occur in estuaries in small numbers and are not dependent on them.

The current view is that many marine fish are estuarine-dependent during their juvenile phase but return to the sea to breed (Potter et al. 1990; 2015; Elliot et al. 2007). The ponds offer sheltered conditions and nutrient-rich environments than can be used by these fish species as nursery areas equivalent to closed estuaries (Whitfield 1994). The fish inhabiting the mining ponds do seem capable of reproducing in the ponds as (a) gonads were recorded in all stages of development, and (b) there was evidence of juvenile mullet and juvenile steenbras in at least some of the ponds, even although the ponds were isolated from the sea. This is of course not conclusive evidence, as they might have been introduced by wash-overs from the sea. However, individuals of relatively young ages (1-6 y) were recorded in the north ponds, which have been isolated from the sea for periods well in excess of this number of years, so that recruitment from the sea is unlikely in those ponds.

The mining ponds can play an important role as nurseries, as they have characteristics that are advantageous for juveniles, such as high temperatures and refuge from predators, which may respectively increase growth rates and survival (Potter et al. 2015). However, the ponds supported only a small set of marine fish species: although not substantially less than the numbers of marine species recorded in other west-coast estuaries (Harrison 1988; Whitfield 2005; Lamberth 2013).

The sheltered conditions of the mining ponds allow steenbras, which has a slow and prolonged growth rate, to grow faster, particularly in the south and middle ponds where a diversity of macroinvertebrates is available as food, although the steenbras did not achieve sizes as great as in the sea. However, any changes in the salinity levels within the

ponds will result in changes in the variety and types of food organisms present (Forbes and Cyrus 1993; Whitfield and Bate 2007), with likely effects on fish growth.

All marine organisms have a range of tolerances to salinity, and fish trapped inside the ponds have to be able to adjust and adapt to changes in salinity to survive. Fish are unlikely to survive if salinities approach lethal levels or exceed their tolerance ranges, or their condition may decline even if they do survive. As outlined in Chapter 1 the regression of salinity against age indicates that pond salinities are likely to remain tolerable for about 10 years, but after that, they will climb steeply to stressful levels above 70‰, causing osmoregulatory stresses in marine organisms, and particularly for fish trapped inside the mining ponds (Whitfield et al. 2006).

In the short term, once a pond is dewatered and the area is mined out, the entire wetland ecosystem is destroyed, although recovery can take place once the ponds begin to fill again. In the longer term, biota in ponds older than >15 years will experience physical stresses once salinity levels rise to dangerous levels. Therefore, the potential for fish to survive in the ponds will depend on (a) the possibility of preserving selected ponds (perhaps where aquaculture can develop to contribute to the development of sustainable post-mining economic activities), (b) understanding the benthic invertebrate faunal availability, which is important for interpreting the health and survival of fish and (c) the long-term viability of the ponds in the light that they are likely to become shallow and hypersaline after about 15 years, after which mullet – which can tolerate salinities up to 90‰ (Whitfield 2019) – will likely be the sole surviving fish.

Returning to the hypotheses advanced in the Introduction, my general hypothesis was that the ponds will '*support and provide habitat for marine fish*'. This was upheld by the fact that populations of marine species are well established in the ponds, spanning multiple age groups, and seemingly being capable of reproduction within the ponds. However, the number of species caught regularly was very small, even by the standards of the generally low-diversity estuaries in the region (Harrison 1998; Lamberth 2013).

In abstracted form, specific hypotheses were:

1) *There would be significant differences in community composition among areas … and the diversity and relative amounts of fish would be least in the north ponds*. In reality, no difference in community composition emerged. All ponds had just two dominant species that accounted for ca. 95% of the fish, i.e., the west coast steenbras *Lithognathus aureti* and the southern mullet *Chelon richardsonii*. In surveys of west-coast estuaries, Harrison (1998) recorded *C. richardsonii* as being dominant in all cases, constituting 59.6 to 94.3% on the numbers of fish caught. This is unsurprising given the wide salinity tolerance of the species (2-90‰ – Whitfield 1996). Numbers of this mullet and of steenbras were highest in north ponds, despite the high salinities there. There was no correlation between salinity and total fish numbers or numbers of mullet; but there was a negative exponential relationship between salinity and numbers of steenbras.

2) *Mean sizes would be greatest in the north … recruits would be absent from north ponds … and body condition would be lowest in the north ponds for steenbras and highest for mullet in middle ponds*. Sizes of steenbras and mullet were large in north ponds and greater than those in south ponds, but not larger than those in the middle ponds. The major cause of differences in size related to the second part of the hypothesis, because the absence of recruits of either species in north ponds boosted mean sizes there. As I hypothesised for steenbras, body condition was poorest in the north ponds where benthic invertebrate food was limited. Mullet, being detrital and planktonic feeders (Masson and Marais 1975; Marais 1980) were not expected to suffer from food shortage in the north ponds, as there was no reason to suspect that detrital material would be low there. Based on higher levels of chlorophyll-*a* in the middle ponds, I suspected that body condition of mullet would be better there, and that was indeed the case. However, statistical analysis failed to detect significant differences among areas in the concentrations of chlorophyll-*a*, so not much weight can be placed on the apparent relationship between high chl-*a* and good body condition in the middle ponds.

3) *Steenbras would have the smallest range of diet and the lowest Gut Fullness Indices (GFIs) in the north*. This prediction was based on the diversity of food sources in the different areas, and the likely stress imposed on fish in the north ponds. Qualitative comparisons of the stomach contents of steenbras revealed they did have a more constrained range of diet in the north, but gut fullness, while highest in the south, was

lowest in middle ponds. Dietary diversity could not be assessed for mullet because their diet uniformly comprised just diatoms and detritus, but they did adhere to the prediction that GFI would be lowest in the north.

4) *The Gonadosomatic Index (GSI) would be lowest for steenbras in the north ponds.* Analysis of the data upheld this hypothesis, but the differences distilled to a single very high value for GSI for the middle-plus-south ponds in June, resulting in an interaction between area and month. It is also possible that GSI in the north ponds was elevated simply because those ponds were never connected to the sea, so that the fish could not escape to spawn at sea.
5) *Growth rates and size structures would be the same in ponds as in the sea.* Comparisons were feasible only for *L. aureti*, for which there are detailed studies of sea-based fish, as outlined above. No clear differences emerged between the growth rates I obtained in the three areas and in the sea, but small sample sizes in some cases prevented rigorous statistical comparisons. Pooling my data for the ponds did, however, show that growth in the ponds was faster than the rates determined by Beyer et al. (1999) and Holtzhausen and Kirchner (2001) for the adjacent southern marine population of *L. aureti*, but slower than that of the northern marine population.

Integrating this information, the diamond-mining ponds do provide habitat for marine fish, but only for a small number of species, dominated by the west coast steenbras *L. aureti* and the southern mullet *C. richardsonii*. Both species survived for prolonged periods in the ponds, reaching ages of respectively >25 y and 18 y, even in ponds in the north that became hypersaline. Increased salinity did not diminish the numbers of fish, or the numbers of mullet, but salinity was negatively related to steenbras numbers. Unexpectedly, community composition was not different among areas, although diversity was lower in the north ponds. There was circumstantial evidence that both of the dominant species may breed in the ponds, based mainly on the existence of small (young) size classes in ponds that have been isolated from the sea for periods longer that the age of the fish. Steenbras exhibited all stages of sexual reproduction in the south and middle ponds but not in the north ponds, where the population predominantly comprised old, large females, with males and individuals in a protandrous state being virtually absent, raising the spectre that fertilisation will be unlikely in the future.

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APPENDICES

Appendix 4.1: Physical data for the ponds surveyed for ichthyofauna. Data derived from Chapter 1, with addition of data on fish numbers from this chapter.

Pond	Temp. (°C)	Salinity (‰)	pH	Chl-*a* (µg/L)	DO (%)	South African Mullet	West Coast Steenbras	Total number of fish
S3	20.07	37.65	8.15	8	102.9	31	25	56
S16	21.01	37.76	8.19	6.6	112.9	11	15	32
S17	22.13	38.02	8.09	3.6	103.6	0	17	17
S20	18.75	40.58	7.9	26.7	94.2	0	2	2
S23	22.6	39.66	8.31	5.6	105.2	0	2	3
M25	20.3	39.21	8.17	17.6	105.7	17	0	17
M26	18.67	39.29	8.16	72	95.6	0	2	2
M27	19.02	35.99	8.05	12.5	98.8	20	0	20
M28	16.76	35.55	7.91	24.2	97.7	5	25	30
M39	14.31	35.28	8.13	14	108	0	3	3
N44	21.91	70.77	8.24	2.1	109.2	0	1	1
N49	21.43	70.79	8.21	8.1	102.4	16	63	79
N102	23.33	78.19	8	11.2	120	31	0	31
N109	24.97	88.97	8.66	15.6	136.5	9	0	9
N116	24.16	85.05	8.26	5.4	101	30	0	30

CHAPTER 5: Synthesis

The objective of this study was to determine the ecological value of the mining ponds that are generated by the process of diamond mining and create a unique habitat on a coast in which this wetland habitat is largely absent because of the arid nature of the environment. The study area lies within Southern Coastal Mines, a high security area surrounded by a double fence where public access is strictly controlled. One of the biggest limitations in understanding the ecological role of these mining ponds is due to the limited number of studies conducted in the area because of these access restrictions. My study focused on the viability of the ponds and adopted an ecosystem approach and provided detailed case studies of the selected mining ponds with a focus on the extent to which they serve as a useful function as ecosystems in terms of their physical properties, saltmarsh distribution, bird conservation and fish colonisation.

In Chapter 1, the physical properties of the ponds were monitored which gave valuable insights into the physical conditions and availability of resources that support ecological systems and sustain the biota present within the ponds. Several trends emerged from the measurements of the physical properties of the representative ponds and the adjacent ocean water, of which significant effects were expected, given the low inputs of freshwater, the evaporation anticipated in this arid region and due to the construction of seawalls that cut off the ponds from the influences of the ocean to varying degrees, and variations in the ages of ponds, some of which are over 30 years old.

Temperature was higher in the ponds than the sea, higher in the north ponds than middle or south ponds, and increased by day but dropped by night. Salinity in the north ponds was significantly elevated over that in the remaining ponds, which approximated that of the open sea. For ponds less than 10 years old, salinity levels remained close to those of seawater; but thereafter (notably in the north ponds) values rose steeply to as much as 115‰, exceeding the tolerance levels of most estuarine and marine species (Whitfield et al. 1981).

Measures of pH and dissolved oxygen did not differ significantly among the pond areas but values were above those of the sea. Chlorophyll-*a* levels were greater in middle than

north ponds but no other significant differences existed among areas. These results led to the conclusion that the physical conditions of the mining ponds are such that they can serve as a functional equivalent to natural closed estuaries, but that they have a finite life before salinities rise to ecologically stressful levels.

Chapter 2 describes how the ponds offer an environment for the establishment, growth and distribution of a single saltmarsh species, *Salicornia natalensis*, which benefits from the presence of the ponds. The orientation of saltmarshes around the ponds was correlated with the direction of prevailing winds, suggesting that wind dispersal is responsible for the dispersal of the plants along the coast. The proportion of ponds with saltmarshes was greatest in the north, as expected from their greater age and longer absence of mining activities. The health of the saltmarshes was greatest in the south, where the ponds are relatively young and their salinity levels approximate to those of seawater due to seepage across the seawalls. However, these communities will be disturbed by future mining activities. In addition, once mining ends, saltmarshes will become stressed due to higher salinities as ponds age, which may cause the sediment electrical conductivity to exceed the tolerance limit for *S. natalensis* (Shaw 2007). The ponds may, however, serve as 'stepping stones' in the dispersal of *S. natalensis* along the coast, which is otherwise devoid of habitat suitable for the species.

Chapter 3 provided an overview of the bird species that occupy the mining ponds. The ponds supported 36 species including ten regional endemics, six regionally endangered species and nine species that exceed thresholds specified for Important Bird Areas (IBAs), regional IBAs, 1% of the southern African population, or 1% of the flyway population. There were significant differences in the community compositions and relative abundances among the ponds in the north, middle and south areas.

Bird numbers were lowest (around 62 per pond) in ponds that were being actively mined and disturbance was high, but rose to levels 6-10 times greater post-mining, even in the high-salinity north ponds. Greatest numbers (averaging 1266 birds per pond) were associated with periods when the ponds were 'dewatered', exposing benthic food for the birds.

There was no correlation between salinity and bird numbers and the life of the ponds as a bird habitat may extend beyond the time they begin to experience elevated salinities, possibly because the birds have access to drinking water from other sources such as the Orange River. Similarly, the absence of any negative effect of pond age on bird numbers or species richness suggests that even very old ponds will remain a viable habitat for birds. The near threatened Blacknecked Grebe (*Podiceps nigricollis*), endangered Cape Cormorant (*Phalacrocorax capensis*) and Common Tern (*Sterna hirundo*) were among the most abundant species in the study area. The ponds are an important habitat for these species especially since the decline in Cape Cormorant and Common Tern numbers along the west coast in general, and at the Orange River Ramsar site in particular (Anderson et al. 2003; Kemper et al. 2007; Crawford et al. 2007, 2018).

Overall, the mining ponds can be regarded as an Important Bird Area, even though the total number of birds supported falls short of the Ramsar threshold of 20000 birds and the ponds constitute an attractive alternative habitat for waterbirds to nest, forage and roost (Velasquez 1993; Ma et al. 2009; Chokri and Selmi 2011).

Chapter 4 dealt with the ecological importance of the mining ponds in terms of their occupation by marine fish. The ponds offer sheltered conditions and nutrient-enriched environments that can be used by both adult fish and juveniles occupying them as nursery areas fulfilling a role akin to that of closed estuaries (Whitfield 1994). The marine ponds were colonised mainly by west coast steenbras *Lithognathus aureti*, southern mullet *Chelon richardsonii* and to a lesser extent white stumpnose *Rhabdosargus globiceps* and silver kob *Argyrosomus inodorus*. The dominant species, west coast steenbras and southern mullet, both endemic to the southern African coast, are important commercial and recreational species in Namibia, spawning at sea and entering estuaries as juveniles (O'Toole 1997; Bianchi et al. 1999; Boyer and Hampton 2001; Holtzhausen et al. 2001; Elliot et al. 2007). The number of species caught regularly was very small, even by the standards of the generally low-diversity estuaries in the region (Harrison 1998; Lamberth 2013). Nevertheless, for this small set of marine fish species, the mining ponds can play an important role as nurseries, as they have characteristics that are advantageous for juveniles, such as high temperatures and refuge from predators, which may increase growth rates and survival (Potter et al. 2015). However, for a nursery

function to be fulfilled, the marine fish must be able to return to the sea to complete their life cycle, otherwise there is a danger that the ponds may become 'coastal traps'. Once mining comes to an end, the seawalls will no longer be maintained and breaching of ponds will establish direct contact with the sea and restore salinities to levels comparable with the sea.

No difference in community composition emerged among ponds in the north, middle and south, with ponds having just two dominant species (west coast steenbras and southern mullet) that accounted for ca. 95% of the fish. There was no correlation between salinity and total fish numbers or numbers of mullet; but there was a negative exponential relationship between salinity and numbers of steenbras.

The ponds support and provide habitat for marine fish spanning multiple age groups, implying that the fish are capable of reproducing in the ponds. Steenbras exhibited all stages of sexual reproduction in the south but not the north ponds, where the population predominantly comprised old, large females, with males and individuals in a protandrous state being virtually absent, so that fertilisation will be unlikely in the future in the north ponds. The occurrence of juveniles in south ponds, some of which had been isolated from the sea for prolonged periods, supports the idea that the fish were breeding there.

Sizes of steenbras and mullet were large in north ponds and greater than those in south ponds, but not larger than those in the middle ponds. Steenbras body condition was poorest in the north ponds where benthic invertebrate food was limited. Mullet, being detrital and planktonic feeders (Masson and Marais 1975; Marais 1980) were not expected to suffer from food shortage in the north ponds, as there was no reason to suspect that detrital material would be low there. Their body condition was highest in the middle ponds, coinciding with the high chlorophyll-*a* levels there, which may elevate benthic detrital food for the mullet. The stomach contents of steenbras revealed they did have a more constrained range of diet in the north, but gut fullness, while highest in the south, was lowest in middle ponds. The diversity of food sources was smaller in the north.

Management recommendations

The presence of saltmarshes, large numbers of birds, and a limited set of fish in the mining ponds reflects the largely favourable physical conditions that permit the coastal ponds to serve many of the roles of 'closed estuaries' in this otherwise arid environment. The potential for the ponds to continue to be used as alternative habitats for the distribution of saltmarshes and supporting of bird and fish species depend on:

a. The possibility of preserving selected ponds for various functions, including (i) their use as a conservation tool, (ii) development of aquaculture and tourism to contribute to the development of sustainable post-mining economic activities, and (iii) enhancement of avian conservation by the creation of artificial islands in selected ponds after mining, for birds nesting and roosting sites.
b. The long-term ecological viability of the ponds depends on the fact that they are likely to become shallow and hypersaline after about 15 years. While mullet – which can tolerate salinities up to 90‰ (Whitfield 2019) – may continue to survive there, this is far less likely for steenbras. The life of the ponds as a bird habitat, however, may extend beyond the time they begin to experience elevated salinities.
c. Understanding whether the benthic invertebrate fauna is sufficient to support viable bird and fish populations in the ponds, which is important to interpreting future changes in bird abundance and occurrence (Colwell and Taft 2000), and the health and survival of fish. Changes in the physical characteristics of the ponds caused by mining have immediate consequences for birds. There is a need to continue monitoring bird populations and determine conservation value assessments in the restricted diamond area to track changes. For example, as one pond is disturbed by mining, another site may provide permanent water as a safe refuge site. This will improve understanding of how ponds in different areas continue to serve different purposes – with some ponds providing a safe place to breed and others providing better foraging opportunities, as is the case when ponds are 'dewatered'.

Returning to the original question of whether ponds serve as viable ecosystems, the answers must be a qualified 'yes'. Restoration of the ponds to a condition approximating circumstance before mining began would eliminate the wetland role that the ponds fulfil. The case for retaining the ponds is perhaps strongest in terms of their role as habitat for

wetland birds, as the ponds support significant numbers and a high diversity of birds, and neither their numbers nor their diversity seem to be diminished by rising salinities as the ponds age.

For fish, the ponds are occupied by marine fish, and there is at least circumstantial evidence that they breed in the ponds and can grow at rates that are comparable to (or greater than) those in the sea; but the diversity of fish in the ponds is very low, and the long-term effect of mounting salinities will limit the life-of-ponds as a fish habitat. Moreover, if the pond become cut off from the sea, exchange between the sea ponds will cease.

For the saltmarshes associated with the ponds, a single species, *Salicornia natalensis*, benefits from the ponds and forms substantial areas around them. The probability that dispersal among ponds is wind-driven means that the ponds do act as 'stepping stones' along the otherwise arid and water-deprived coast. The ecological significance of this is, however, limited by the fact that only one species benefits in this way, coupled with the fact that stress levels for the saltmarsh will progressively increase as salinities rise.

Certainly the ponds remain ecologically viable – and remain so for surprisingly prolonged periods of time. While they cannot be regarded as natural systems, their biotic composition and functioning are remarkably like those of temporary open/closed estuaries during their closed phase (Froneman 2018). Their main limitations are restricted access to the sea, and the aridity of the area, which leads to a virtual absence of freshwater input – features that Whitfield and Bate (2007) and Whitfield et al. (2012) argue are pivotal to the health of these systems. The fact that periodic breaches and overtopping of seawater into the ponds take place in situations where the ponds lie sufficiently close to the sea does, however, maintain contact between the ponds and the sea.

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